Stand-Level Response to Sanitation Thinning in Quaking Aspen Exclosures

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A Professional Paper
Submitted in Partial Fulfillment
of the Requirements for the Degree of
Master Of Forestry

School of Forestry
Northern Arizona University
May 5th, 2026

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Abstract

Quaking aspen (*Populus tremuloides*) is a culturally, aesthetically, and ecologically important species in the U.S. Southwest, yet many populations lack resilience to changing disturbance regimes and a warming climate. Current aspen decline is driven by warming temperatures, drought, ungulate browsing, and biotic agents, including oystershell scale (OSS; *Lepidosaphes ulmi*), an invasive sapsucking insect causing severe dieback and mortality. Silvicultural treatments are commonly used to improve resilience; however, land managers lack proven strategies to address OSS outbreaks, with little known about the effectiveness of sanitation thinning (the removal of infested or declining trees) in infested aspen stands. We evaluated whether sanitation thinning improves residual overstory growth and tree vigor in aspen stands affected by OSS. We collected tree metrics and increment cores to quantify overstory growth and crown condition. Three-years following sanitation thinning, we found increases in annual radial growth rates, larger crown ratios, and less crown dieback, and regeneration densities in treated units, indicating improved tree condition and stand structure. In contrast, untreated units did not show significant differences in radial growth and regeneration densities, nor demonstrated improved crown condition, over the same study period. Additionally, OSS-infested stems from untreated units showed decreased growth rates, showing continued decline. Findings support sanitation thinning as a management strategy for improving resilience in Southwestern aspen stands impacted by multiple stressors.

Professional Ethics Statement

My professional code of ethics strongly aligns with the principles of the Society of American Foresters (SAF) Code of Ethics, particularly its emphasis on continuous forest stewardship, upholding scientific integrity, and carrying the responsibility of service.

I recognize the necessity to manage forests for both current and future generations, ensuring the long-term capacity of these ecosystems to provide numerous significant ecological and social values. To do this, foresters must rely on and contribute to sound science, pledge to use their abilities to help formulate sound forest policies and laws, and provide open and honest communication.

In the context of this professional paper and my research on quaking aspen silvicultural treatments, these principles shaped the motive behind the research and determined how I interpret and present results. I see it as my responsibility to offer sound research and to present these findings clearly, with the ultimate purpose of supporting aspen persistence under changing conditions while simultaneously balancing landowner objectives and broader ecological considerations. This includes acknowledging previous contributions of other professionals, utilizing the best available science, being transparent about uncertainty, and ensuring communication with land managers and stakeholders, with the purpose to contribute to informed decisions around forest management and policy.

Sincerely,
Jackson Bledsoe

Acknowledgements

Without the contributions, support, and assistance from others, none of this work would have been possible.

Thank you to Kristen Waring, my advisor, for the countless hours offered and the numerous questions answered on my behalf. Despite this experience being quite the undertaking, this experience has radically changed the trajectory of my career, and I cannot thank you enough!

Thank you to my support system, my mother and partner, for always being available and cheering me on. Additionally, thanks to the members of the Silviculture and Forest Health for the camaraderie and making computer work a little less dull.

Thank you to Abiral Acharya, Marcos Arbizo, Chander Craft, Bear Murillo, and Kristen Waring for the dedicating their days off to helping in the field and sharing the arm pain from increment coring.

Thank you to my professional paper readers, Dr. Dave Auty and Mark Nabel, for all their time and feedback throughout this process.

I truly appreciate each one of you for the role you have played in my journey!

Sincerely,
Jackson Bledsoe

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Introduction

Background On Southwestern Aspen Ecosystems

Quaking aspen (*Populus tremuloides* Michx.) is the most widely distributed tree in North America (Little, 1971; Perala, 1985). Aspen populations in the Southwestern United States sit near the southern edge of the range and have been experiencing compounding pressure from extreme drought, chronic ungulate browsing, and wildfire for at least two decades (Zegler et al., 2012; Crouch et al., 2023).

Quaking aspen are unique clonal organisms with an underground root system up to 100 acres or more in size that mainly vegetatively reproduces individual stems of identical genetics (Barnes, 1966; Perala, 1985). Regenerating individuals, called "root suckers," arise from adventitious sprouts on shallow lateral root systems (Barnes, 1966). Disturbance events, such as fire or cutting, stimulate a suckering response with immediate regeneration observed in densities of up to 55,000 stems per acre and height growth up to five feet or more in the first year (Perala, 1985; Crouch et al., 2025).

Quaking aspen ecosystems have disproportionate significance in the Southwest due to their strong ecological, economic, cultural, and aesthetic value (Chong et al., 2001; McCool, 2001). Ecologically, aspen forests support biodiversity by providing habitat for a wide range of species and by maintaining important ecosystem processes such as carbon storage and watershed stability (Shepperd et al., 2006, Rogers et al., 2020). Local to the Southwest, aspen serves as a year-round food resource for ruffed grouse *(Bonasa umbellus)* and provides browse (branches, twigs, young sprouts) for deer and elk (Perala, 1985).

Aspens also have significant importance in society. Economically, aspen is an important and valuable part in sawtimber and pulpwood national resources (Perala, 1985; McCool, 2001), although this value is limited in the Southwestern United States. Furthermore, they contribute to local economies by supporting outdoor recreation and tourism (Assal, 2020): Between 2017-2018, Flagstaff, Arizona tourism included over 5.2 million visitors with 32% participating in recreation on local trails (*2017-2018 Flagstaff Visitor Study*, 2019), with viewing fall changing colors of aspens being a noteworthy portion (U.S. Forest Service, 2018). Culturally and symbolically, aspen ecosystems hold high intangible value and are spiritually important to Indigenous communities and local traditions (McCool, 2001).

Quaking aspen stands are significantly more likely to reduce fire occurrence, behavior, and severity (Nesbit et al., 2023; Harris et al., 2025), and as a result, aspen stands can function as natural firebreaks. In wildland-urban interface areas where limited fire intensity and rate of spread are critical for protecting communities, the presence or restoration of aspen can help contribute to landscape-level resilience.

Aspen Stand Dynamics

Quaking aspen is a disturbance-dependent species historically maintained by stand-replacing events, primarily wildfire, which stimulate widespread root suckering and promote even-aged stand development (Perala, 1985; Crouch et al., 2023). Aspen in western North America occur in functional types: either as stable populations that maintain stand dominance under conditions unfavorable for conifer establishment, or seral stands, which aspen initially dominate from disturbance but are slowly replaced by conifers (Rogers et al., 2014). A population's functional type cannot readily be determined as knowing structure and composition over multiple ecological rotations is necessary (Rogers et al., 2014). A third functional type has been proposed, describing decadent and mature aspen stems with little to no regeneration, where replacement is inevitable without management intervention (Bartos, 2001). Identifying the functional type of Southwestern aspen stands prior to management is crucial, as many aspen populations have recently regenerated under previous management interventions, altered disturbance regimes, and shifted stand dynamics (Crouch et al., 2023).

Changes in forest structure following disturbance in portions of a landscape is not necessarily a forest health concern, yet large-scale changes in successional processes outside of normal variation over a long historical period can be of concern when managing for forest health and resilience (Kolb et al., 1994). Under historic conditions, native insects and pathogens are a natural component of these ecosystems, typically acting as secondary agents that affect weakened trees, but would not normally attack vigorous trees (Marchetti et al., 2011).

Aspen populations in the Southwestern United States have lately experienced increasing shifts in stand structure driven by a combination of climatic stress, altered disturbance regimes, and biotic pressures (Worrall et al., 2008; Zegler et al., 2012; Stoddard et al., 2024; Crouch et al., 2025). Environmental stressors can compound one another, contributing to widespread decline, reduced tree vigor, and inconsistent regeneration (Marchetti et al., 2011). Widespread mortality events have been observed at the stand and landscape scales across the Southwest (Zegler et al., 2012). Compounding biotic factors, including many fungal pathogens and insects, represent the largest cause of mortality (Perala, 1985; Zegler et al., 2012). This causes concern for long-term stand persistence, as altered stand structures and disturbances regimes can limit the establishment of regeneration and recruitment of stems to ensure overstory replacement (Rogers et al., 2010; Zegler et al., 2012), a process can especially be limited on lower elevation and more exposed sites (Stoddard et al., 2024).

Climatic stress and ungulate browsing further contribute to regeneration failure in Southwestern aspen ecosystems. Prolonged drought and increased temperatures reduce tree vigor and increase susceptibility to mortality (Worrall et al., 2008). Additionally, high abundance of ungulates, primarily elk and deer, can heavily browse regenerating suckers, preventing stems from growing beyond browse height and recruiting into overstory size classes (Figure 1; Perala, 1985; Kay & Bartos, 2000; Crouch et al., 2023). Persistent browsing pressure over several years can exhaust energy supply and limit sucker production (Rolf, 2001; Frey et al., 2003)

Figure 1. An example of extreme ungulate browse pressure in an unexclosed quaking aspen stand north of Flagstaff, Arizona, preventing recruitment of regenerating stems and resulting in uniform canopy base heights.

Additional stress has been introduced through altered disturbance regimes and increased competition from conifers. Fire suppression over the past century has reduced the frequency and intensity of stand-replacing fire events, allowing conifers to establish and encroach into aspen stands (Kay, 1997; Nesbit et al., 2023). This increases competition for resources such as light, water, and nutrients, further reducing aspen growth and limiting regeneration, contributing to long-term stand conversion (Zegler et al., 2012). In combination with aging stands, these altered conditions contribute to declining productivity and increased susceptibility to pathogens and insects (Worrall et al., 2008). Under a seral aspen functional type, less than adequate regeneration and recruitment can contribute to stand elimination or type conversion (Rogers et al., 2014).

Oystershell scale (Figure 2; OSS; *Lepidosaphes ulmi* Linn.) is an invasive, armored, generalist scale insect that poses a threat to almost 130 species of trees and shrubs (Samarasinghe, 1965). Within the past few decades, OSS has spread into aspen stands across northern Arizona, causing outbreaks and subsequent heavy crown dieback and mortality (Grady, 2017). As a sap-sucking insect, OSS can contribute to declines in tree vigor through high bark and branch infestation densities, causing extensive dieback and disrupting vascular function (Samarasinghe, 1965). Climate is the most important factor driving OSS invasions of aspen ecosystems (Crouch et al., 2024).

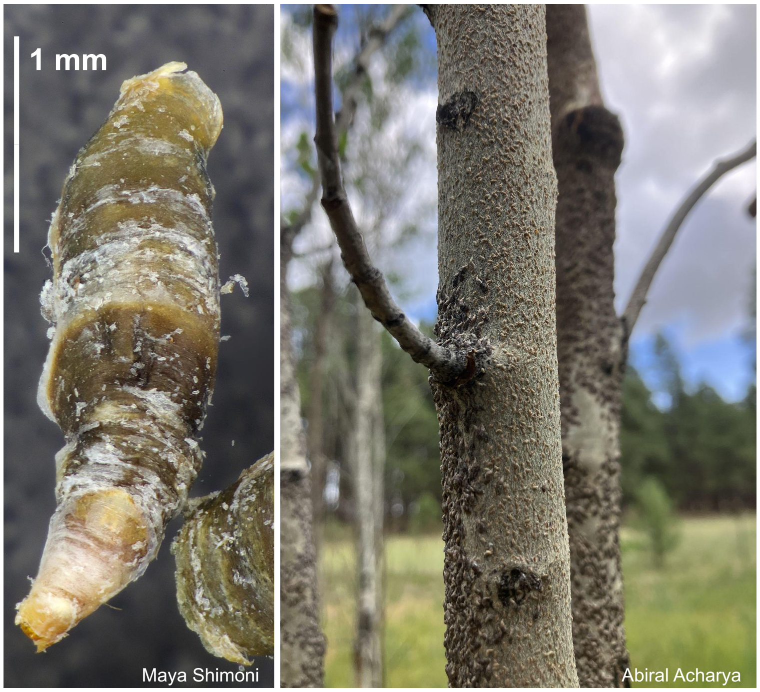

Figure 2. Oystershell scale (OSS: *Lepidosaphes ulmi* Linn.) individual (left, photo by Maya Shimoni) and an OSS population of all life stages (right, photo by Abiral Acharya) present on the entire surface area of the bole pictured.

OSS may also impact stand dynamics by affecting regeneration and intermediate-sized stems. Mortality of these size classes limits recruitment into the overstory, further threatening the long-term stand persistence (Crouch et al., 2025). Compounding the effects of recent and severe interacting stressors, including increasing temperatures, less available moisture, ungulate browsing, and competition, OSS outbreaks contribute to reduced regeneration success and ongoing aspen decline in the region (Crouch et al., 2023).

Stand Management

Recently, managing forest stands for “resilience” in the presence of insects, diseases, and drought, has become a priority. Resilience is the ability of a forest ecosystem to tolerate disturbance while retaining the same ecosystem function (Walker et al., 2004). At the stand level, individual tree vigor is essential for supporting overall resilience (Byrant et al., 2019), however, tree vigor is a broad term that is difficult to be directly measured.

Aspens are currently being managed as a “sensitive resource” (U.S. Forest Service, 2018). Declines in quaking aspen across the Southwestern United States have led to an increased need for active and targeted management (Rogers et al., 2014; Crouch et al., 2024). In the absence of frequent natural disturbances that historically maintained aspen dominance, land managers increasingly rely on interventions to increase resilience by promoting regeneration, reducing competition, and improving stand conditions. However, the effectiveness of these approaches can vary depending on the underlying stand dynamics, disturbance history, and interacting stressors, highlighting the need for ecologically-based management strategies (Rogers et al., 2014).

‘Aspen exclosures,’ enclosed groups of aspen within fences, were constructed across the Southwest to support stand persistence by allowing regeneration to grow above browsing height to avoid herbivory pressure (Kay & Bartos, 2000). Exclosures have been effective in allowing suckers to establish and grow beyond browsing height, however, they do not address other factors contributing to aspen decline such as competition and altered disturbance regimes. With a complete absence of browse disturbance, regeneration densities have increased, further supporting OSS populations and impacting aspen regeneration resource availability (Crouch et al., 2021, 2024).

Intermediate treatments and the promotion of regeneration can be valid silvicultural tools to mitigate the effects of introduced pests (Waring & O’Hara, 2005), with thinning treatments potentially increasing the vigor of residual trees by reducing competition for resources (Chagnon et al., 2025). However, the effectiveness of these silvicultural treatments under OSS and other combined stressors in the Southwestern aspen stands remains uncertain (Crouch et al., 2021, 2025).

This study evaluated how silvicultural sanitation thinning influenced residual overstory and regeneration responses in treated and untreated aspen experimental units. Specifically, we (1) quantified changes in residual overstory radial growth rates (annual basal area increment; BAI) following thinning treatment, (2) assessed changes in residual overstory tree crown condition, and (3) quantified regeneration densities. We evaluated these metrics to quantify early treatment responses, make inferences on overall treatment effectiveness, and determine potential stand resilience, with the potential to inform land managers of feasible silvicultural treatment strategies.

Methods

Given the increasing prevalence and severity of oystershell scale (OSS) infestations within these exclosed aspen stands, there is a growing need to incorporate proactive monitoring of this insect into treatment practices. To address this need, recent efforts have established long-term monitoring networks across aspen stands in northern Arizona, conducting repeated measurements of tree condition, regeneration, and damaging agent presence across a range of treatment types and environmental conditions (Zegler et al., 2012; Crouch et al., 2021, 2024, 2025). This monitoring system provides a foundation for evaluating treatment effectiveness under interacting stressors and for informing adaptive management strategies aimed at increasing aspen resilience under increasing biotic and climatic pressures.

Within this framework, we assessed tree vigor using indicators of crown condition and tree growth, including stem and reproductive changes (Dobbertin, 2005). Crown condition, measured by the ratio of live crown length to total stem height and categorical ratings of branch dieback, reflects foliar growth and crown health, is associated with photosynthetic capacity, involving the ability to capture and utilize resources. Following, stem and reproductive growth, measured from basal area increment and density of regenerating stems, provide measures of resource availability and potential stand structural changes.

Study Area

Four aspen exclosures were selected from the existing permanent plot monitoring network across the Coconino and Kaibab National Forests (NF) in northern Arizona (Figure 3), ranging from 0.7 to 10.0 hectares in size. These exclosures were selected to capture heavy OSS infestations and high tree mortality. Within these exclosures, both treated and untreated units were selected to be a part of this study, although prescriptions differed slightly between forests in terms of residual stems, with size classes targeted on different spacing distances (Figure 3).

On the Coconino NF, two exclosures were selected, each originally a single aspen stand that was later subdivided into multiple treatment units (Figure 4). One exclosure included a sanitation thinned unit treated in 2022 and an untreated control unit (Table 1), while the second contained two sanitation thinning units of identical treatment treated in 2021 and 2022, along with an untreated control unit (Table 1). Treatment specifically entailed sanitation thinning treatments targeted at the removal of declining or OSS-infested stems while retaining the healthiest and least-infested overstory individual at an approximately three square meter spacing (Figure 3). In contrast, untreated units were subjected to no management intervention (Figure 3).

On the Kaibab NF, two paired exclosures were included (Figure 4), one of which received sanitation thinning in 2020, while the other remained untreated (Table 1). The treated exclosure was subjected to a complete coppice of all infested stems under roughly a 20 cm diameter maximum, with no spacing objective established (Figure 3). In contrast, untreated units were also subjected to no management intervention (Figure 3). Kaibab exclosures represented distinct stands with differing structural characteristics (Table 1).

Site conditions differed between forests, with mean elevation approximately 123 m greater on the Coconino sites (Table 1). Climatic differences indicated that the lower elevation exclosures on the Kaibab NF annual temperature averages were 18% warmer and received 6% less precipitation than those on the Coconino National Forest (Table 2).

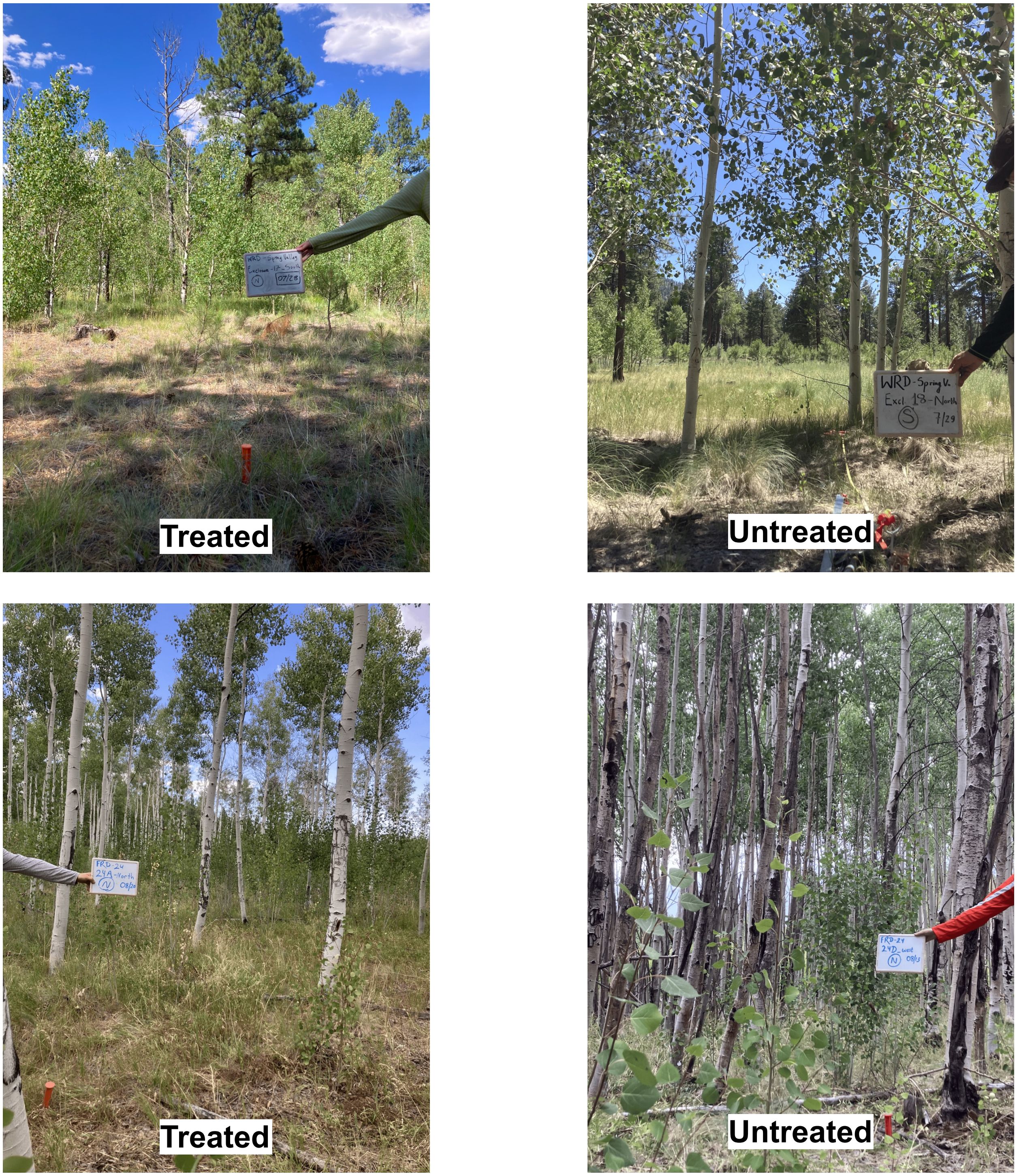

Figure 3. Pictures taken in 2025 from long-term monitoring networks within the study aspen stands. Images highlight differences in stand structure between treated units (left column) and untreated units (right column), across the Kaibab National Forest (top row) and Coconino National Forest (bottom row).

Table 1. Site and unit characteristics for sampled exclosures across Coconino and Kaibab National Forests, including climate, treatment history, and 2025 overstory tree structure. RDI calculated from max SDI = 562 (Long & Shaw, unpublished data)

National Forest	Exclosure Number	Elevation (m)	Mean Slope (%)	MAT * (ºC)	MAP † (mm/yr)	Sampled Treatment Unit	Treatment	Unit Size (ha)	Year Treated	*n* sample trees	Mean Tree Age (yr)	2025 MBAPH ‡ (m2/ha^{-1})	QMD § (cm)	RDI ‖
Coconino	13	2449	2.9	6.9	581	B	sanitation	1.1	2022	14	34	9.5	16.5	0.32
						C	untreated	0.9	-	15	35	15.8	15.8	0.58
	24	2454	3.1	6.9	579	A	sanitation	1.2	2021	10	36	9.6	14.3	0.43
						C		1.1	2022	14	36	10.4	16.9	0.31
						D	untreated	1.1	-	9	36	12.9	17.6	0.38
Kaibab	17	2279	2.1	8.1	548	-	sanitation	1.5	2020	15	23	8.0	17.2	0.25
	18	2270	1.2	8.2	546	internal	untreated	0.7	-	6	47	10.3	32.3	0.09
						external		-	-	9	53	6.9	28.5	0.08

* Mean Annual Temperature
† Mean Annual Precipitation
‡ Mean Basal Area Per Hectare
§ Quadratic Mean Diameter
‖ Relative Density Index (SDI / Max SDI)

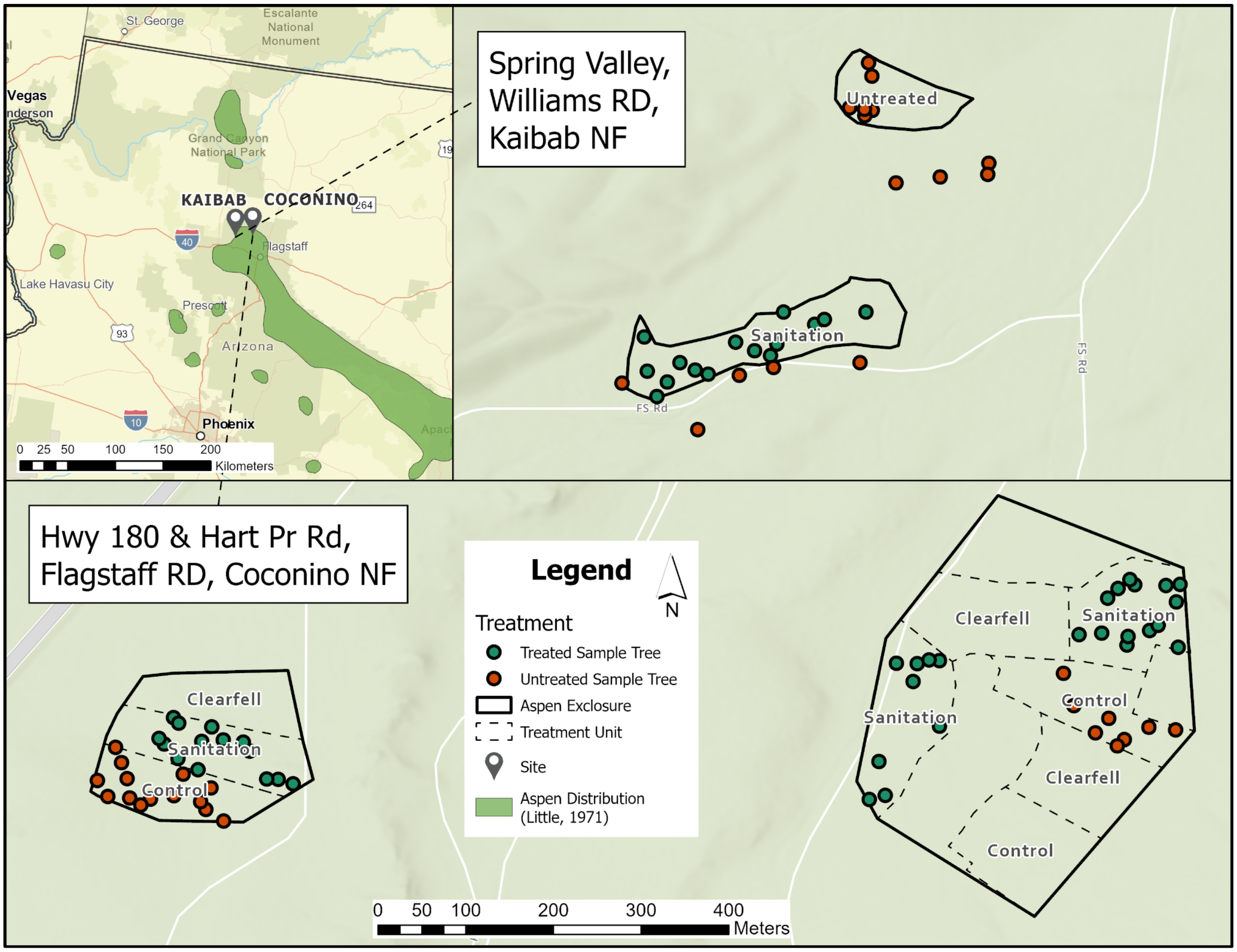

Figure 4. Location of four aspen exclosures across the Coconino and Kaibab National Forests in northern Arizona. Panels show layouts of treatment units within each exclosure. Red points indicate sampled trees. Distances between exclosures were 172 meters apart on Kaibab, and 505 meters apart on Coconino.

Measurements and Sampling

To quantify overstory and regeneration responses, data were collected from two sources: (1) existing regeneration measurements from the permanent monitoring plot network and (2) independently sampled residual overstory trees.

Permanent plots were previously installed within each treatment unit beginning in 2020, with initial measurements collected prior to silvicultural treatment implementation. Following treatment, plots were measured at annual intervals to assess stand development. Regeneration data was obtained from multiple four meter-radius plots established within each treatment unit (plot establishment and data collection procedures described by Crouch et al. (2024)). Although not independently sampled as part of this study, this available data within corresponding treatment units allowed us to quantify regeneration to assess post-treatment responses.

Residual overstory trees were sampled within each treatment unit using a simple random sampling approach. Fifteen randomized GPS points were generated in ESRI ArcGIS Pro within each treatment unit, and at each point, the nearest suitable quaking stem was selected. Sample tree suitability was determined using the following criteria: live, within the overstory size class (> 12.5cm diameter at breast height; DBH), presence of a live crown, and a bole free from major external tissue damage, including cankers, fire damage, or prior OSS scarring. If no suitable tree was present within 50 meters of a generated point, the point was dropped from sampling. Due to limited availability of suitable overstory trees, additional quaking aspen stems outside the exclosure were included (n = 9 stems) at a maximum of 380 meters away from the untreated exclosure to increase sample size on the Kaibab sites (Table 1).

For each measured aspen stem (both overstory and regeneration size classes) basic tree-level data were collected, such as tree condition (live or dead), DBH (cm), and diameter at coring height (cm) which is a necessary piece for measuring basal area increment. Further tree measurement data followed protocol developed in Crouch et al. (2024), including crown dieback (as percentage categories of dead branches within the tree's live crown at 0%, 1-33%, 34-67%, and > 67%), crown ratio (percentage of total tree height occupied by the live crown), and OSS presence and severity ratings (developed by Crouch et al. (2021)).

One increment core was extracted from 40cm aboveground on the uphill side of each overstory stem using a 10mm Haglof increment borer, with effort taken to avoid boring over visible scars, rot, OSS populations, or other observable tissue defects that could affect recent annual growth ring presence. This study does not attempt to correct growth measurements for any tension wood due to the relatively mild slope at sampled sites (Table 1). Collected increment cores were stored in 1-inch PVC tubing with drilled air holes, sealed with tape, and labeled with site name, a unique tree ID, and DBH. Increment cores were allowed to air dry inside of the PVC tubes for a minimum of 48 hours prior to further processing. A total of 93 suitable quaking aspen stems were sampled across treated and untreated units in the Coconino and Kaibab National Forests (Table 1). Of these, 53 stems originated from treated units and 39 from untreated units (Table 1). Sample sizes differed between forests, with Coconino contributing 33 more stems than Kaibab sites.

Sample Processing

Mimicking the increment core sample preparation process of Stokes and Smiley (1968), each core was mounted and sanded after drying with progressively finer grits on an upside-down belt sander (120, 220, 320, 400), followed by hand sanding to 1000 grit.

Following the digitizing and measuring processes of Maxwell & Larsson (2021), sanded cores were then scanned on a flatbed scanner at 1200 dpi and saved as .tif image files. Sample images were then visually crossdated using Cybis CooRecorder 9.8.1 (Larsson, 2023), where calendar years were manually matched to growth rings, while simultaneously measuring ring widths between each identified growth ring. Drought-year location was supported using regional dendrochronological references of ponderosa pine developed by Gardner (2025), allowing for more localized and more recent references than publicly available data, therefore increasing crossdating confidence. Completed measurements were saved as .pos files for each individual core.

Crossdating accuracy was evaluated using partner software Cybis CDendro 9.8.1 (Larsson, 2023). Criteria for acceptance were set at a correlation coefficient >0.5 and a t-test value >5.0 when compared with other samples within treatment units. However, younger trees with short ring-width series (< ~25 years) often fell short of this criterion due to limited temporal overlap to reference chronologies and less extreme marker year presence to increase confidence. In these cases, samples were accepted if visual crossdating matched consistent marker-year alignment and no dating inconsistencies were detected. This adjustment primarily affected stems from a relatively younger stand (overstory stems < 40 years old). Following crossdating verification, all acceptable individual core files were compiled by treatment unit and saved as ring-width series files (.rwl) for further analysis.

Data Analysis

All statistical analyses were completed in R version 4.4.1 (R Core Team, 2024). Plot and increment core sample data were organized using the *dplyr* R package (Wickham et al., 2023), ring-width series data were processed using the *dplR* package (Bunn et al., 2025), and visualizations were created with *gpplot2* (Wickham, 2016). Climatic variation was assessed using averaged climate variables from 1991-2020 obtained from ClimateNA data (Wang et al., 2016). Annual basal area increment (BAI) was calculated for each tree using functions within *dplR* and values were expressed in cm^2yr^{-1}.

For each treatment group, annual mean BAI was calculated by averaging individual tree BAI values within each calendar year. Because treatment implementation occurred in different years, calendar years were standardized relative to treatment year (i.e. pre- and post-treatment periods). Treatment completion varied seasonally; therefore, treatment responses are assessed starting at post-treatment year one.

In addition to radial growth, crown ratio and crown dieback severity were evaluated as indicators of tree health and vigor. Differences in crown ratio, crown dieback, and annual mean BAI differences between treated and untreated groups were evaluated using a Mann-Whitney U test due to non-normal data distributions (Chicco et al., 2025), as determined by Shapiro-Wilk's test of normality ($p < 0.001$). Furthermore, grouping the sample trees by OSS presence allowed us to assess how annual radial growth may be impacted by, or a driver of, infestations outside of any potential treatment responses.

Results

Site Conditions

OSS severity and presence varied across all sites, with variation observed among forest-treatment combinations (Table 2), however, there was no clear relationship with treatment type was clear. Stand mean OSS severity showed a trend of more severe infestations in treated units (Table 2), however, differences were not statistically significant (independent t-tests: $p < 0.001$). Coconino NF treated units showed significantly lower proportions of infested trees than Kaibab NF, (independent two-sided t-test: $p < 0.001$; Table 2), while both Kaibab treated units and overall treated units showed a significantly higher proportion of infested trees than Coconino alone (independent two-sided t-test: $p < 0.001$; Table 2).

Stand structure also differed between study sites. Stand age differed significantly among Kaibab treatment units (independent t-test: $p < 0.001$), comprising a younger treated stand and older untreated stands (Table 2). After sanitation thinning treatment, overstory stem density was 51% lower on the Coconino NF treated units compared to untreated units (Table 1), while Kaibab sites exhibited lower overall densities with no difference between treatments (Table 1). This pattern corresponded with higher mean QMD in the Kaibab untreated unit relative to other site-treatment combinations (Table 2).

Table 2: Site conditions, stand structure, and OSS occurrence across forests compared by treatment status ("U" denotes untreated, "T" denotes treated). Significance of treated unit difference from untreated within a national forest is denoted by "*" ($p < 0.05$).

National Forest Mean	n trees measured		QMD (cm)		Estimated Tree Age (yr)		Crown Ratio (% of total height)		Crown Dieback Severity (%)		OSS Presence (%)		OSS Severity (%)	
	U	T	U	T	U	T	U	T	U	T	U	T	U	T
Coconino	25	38	16.1	16.5	35	35	32.6	39.2*	24.7	17.0	64.0	51.4*	0.40	1.27
Kaibab	15	15	30.1◊	17.3◊	51	23*	50.3	68.3*	41.2	26.7	20.0	93.3*	5.60	14.00
Overall	40	53	23.3	16.7*	41	32*	39.3	47.5*	30.9	19.8*	47.5	63.5*	2.35	4.94

◊ QMD statistical comparison excluded for Kaibab due to lack of replication

Overstory Annual Radial Growth Rates

From the total of 93 increment cores collected, a subset of four increment cores was not included in ring-width analysis. This exclusion was either due to a single occurrence of an issue in identifying pre-treatment rings due to a core defect, or low interseries correlation in increment core crossdating (Table 3). Within-treatment interseries correlations of included increment core samples were above the established minimum criteria (individual core correlation > 0.5), supporting accuracy of crossdating on a stand level.

Table 3: Summary of dendrochronology sampling count and crossdating for increment cores collected from Coconino and Kaibab National Forests across treated and untreated units.

National Forest	**Treatment**	***n* cores collected**	***n* cores used**	**Crossdating Timespan**	**Mean Sample Length (yr)**	**Mean Interseries Correlation**
Coconino	treated	38	38	1988-2025	34.0	0.805
	untreated	25	24	1989-2025	35.0	0.755
Kaibab	treated	15	14	1973-2025	22.9	0.725
	untreated	15	13	1958-2025	45.4	0.724

In the Coconino National Forest, mean BAI (cm^2yr^{-1}) was similar between residual trees in the treated and untreated units during the pre-treatment period (years -5 to -1; Mann-Whitney U tests: $p > 0.05$ for all years). One-year post treatment, no different in radial growth was observed (Mann-Whitney U test: W = 461, $p = 0.94$), however, at two years post-treatment, mean annual BAI was significantly different (Mann-Whitney U test: W = 625, $p = 0.01$; Figure 5).

In the Kaibab National Forest, mean BAI per year (cm^2) differed between treated and untreated units during the pre-treatment period (years -5 to -1), with significant differences observed in all pre-treatment years (Mann-Whitney U tests: $p \leq 0.01$; Figure 5). In contrast, post-treatment radial growth patterns converged between treatments, and no significant differences were observed post-treatment year 0 through year 6 (Mann-Whitney U tests: $p \geq 0.05$). Notably, stems in the Kaibab National Forest site demonstrated greater mean BAI per year, compared with the Coconino National Forest, throughout the study timeframe (Figure 5), although not tested.

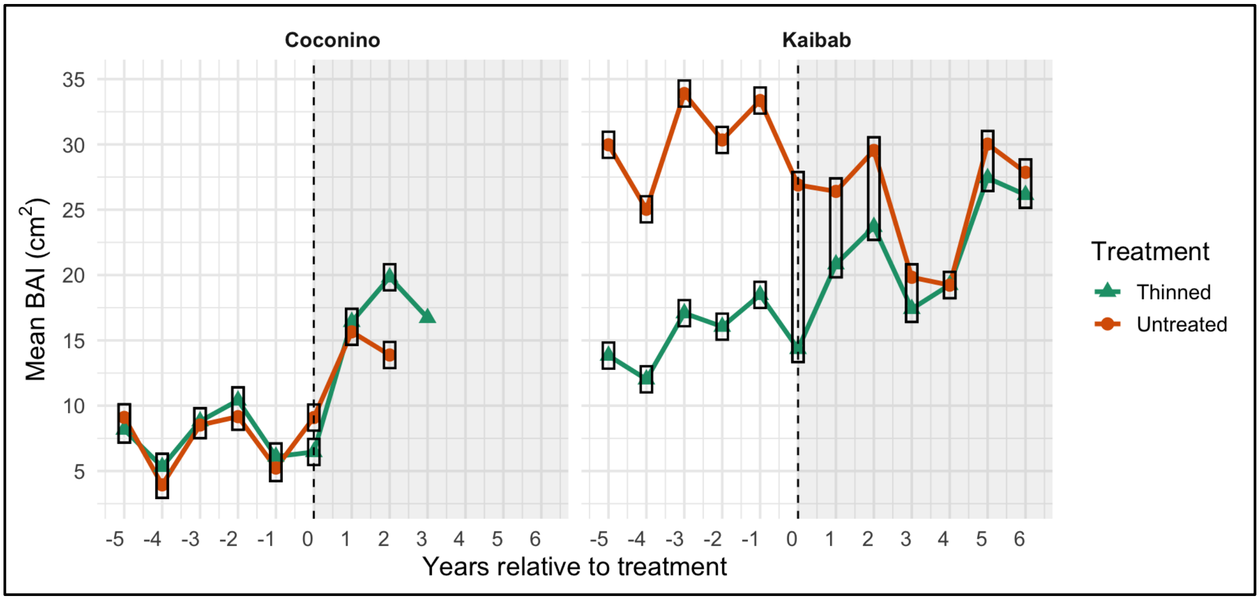

Figure 5. Annual mean basal area increment (cm²) relative to treatment years across treated units and corresponding untreated sites. Separate boxes between treatment types within each year denote significant differences ($p < 0.05$). Year zero represents the calendar year at which silvicultural treatments were completed, however, seasonal timing of treatments varied within calendar years.

Annual mean BAI was calculated from 2000-2024 grouped by OSS infestation presence in 2025. Between 2009 and 2016, stems with OSS exhibited significantly lower mean BAI compared with stems without OSS (Mann-Whitney U tests: $p \leq 0.05$; Figure 6). In contrast, mean BAI did not differ significantly between groups during the periods 2000-2008 and 2017-2024 (Mann-Whitney U tests: $p > 0.05$; Figure 6). However, across the full time series, stems without OSS generally maintained higher mean annual BAI than infested stems (Figure 6).

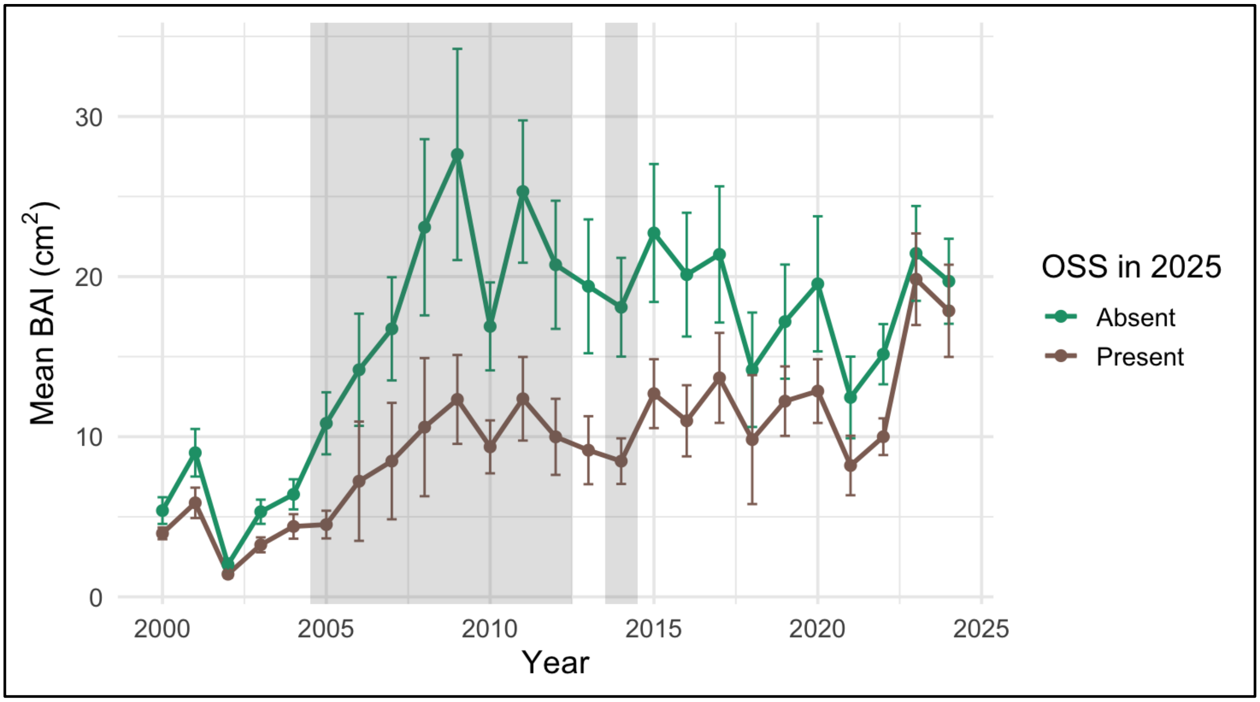

Figure 6. Mean annual basal area increment (BAI; cm²) from 2000 to 2024 for quaking aspen stems across all untreated sites with and without oystershell scale (OSS) infestation observed in 2025. Point values represent mean annual BAI across sampled trees within each group, with error bars (SE = 1) demonstrating range of variation, and connected lines demonstrating year-to-year overall trend. Background shading signifies years where mean BAI were statistically different between stems with and without OSS infestation (Mann-Whitney U tests: $p \leq 0.05$).

Crown Condition Differences

Crown dieback severity showed varying proportions between treated and untreated units (Figure 7). Treated units were dominated by trees in lower dieback categories. In contrast, untreated units contained a greater proportion of trees in moderate and severe dieback categories. Notably, a handful of overstory individuals from each unit did not experience any dieback, with the only exception being the untreated Kaibab exclosure (Figure 7).

In treated units, crown dieback severity was consistently lower than untreated counterpart units study-wide (independent t-test: $t = 2.76$, $p < 0.05$; Table 2), however, no significance was found between treatments in both forests (independent t-test: [Coconino NF, $t = 1.98$, $p = 0.06$];

[Kaibab NF, *t* = 1.74, p = 0.09]; Table 2). The majority of trees from treated units fell within the none to low dieback severity categories, representing 13.2% and 78.9% of observed trees on the Coconino and 6.7% and 60.0% of observed trees on the Kaibab, respectively (Figure 7).

In contrast, untreated units displayed a greater crown dieback severity percentage (Table 2) and higher proportions of trees in moderate and severe dieback categories. Higher-dieback categories (moderate and severe) represented 28.0% of observed trees on the Coconino and 53.3% on the Kaibab, although trees with severe dieback were not observed on the Coconino.

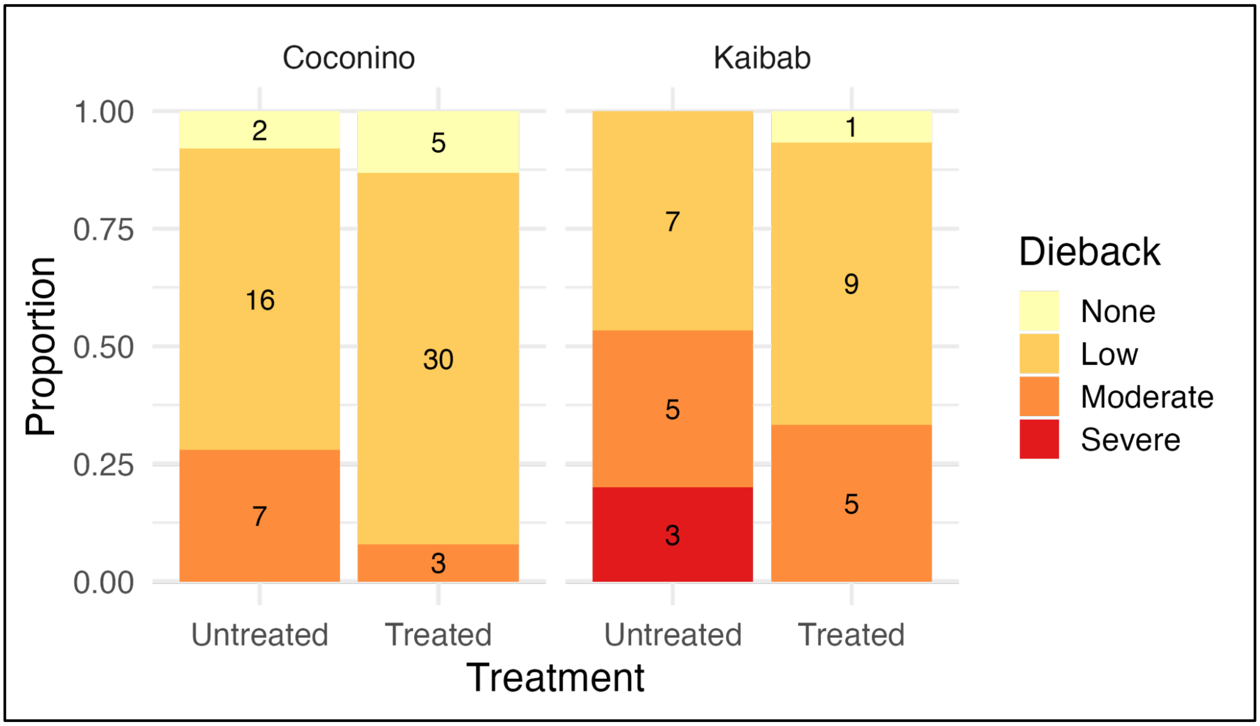

Figure 7. Distribution of crown dieback severity classes for quaking aspen in treated and untreated units across the Coconino and Kaibab National Forests. Dieback severity was categorized as none, low, moderate, or severe, with the number of observed trees of each category labeled over corresponding bars.

Crown ratio differed between treatment groups within both forests, with treated units showing on average longer crown ratios than untreated units (independent t-test: $t = -2.02$, $p < 0.05$; Table 2). On the Coconino, differences lay on a narrower distribution, with crown ratios

20% greater compared with trees from untreated units (Figure 8). On the Kaibab, differences were more pronounced, where trees from treated units showed 36% higher crown ratios, yet a broader distribution into higher values (Figure 8). Mean crown ratio also differed between forests, with trees from Kaibab treated units demonstrating 74% greater crown lengths than Coconino treated units (Figure 8).

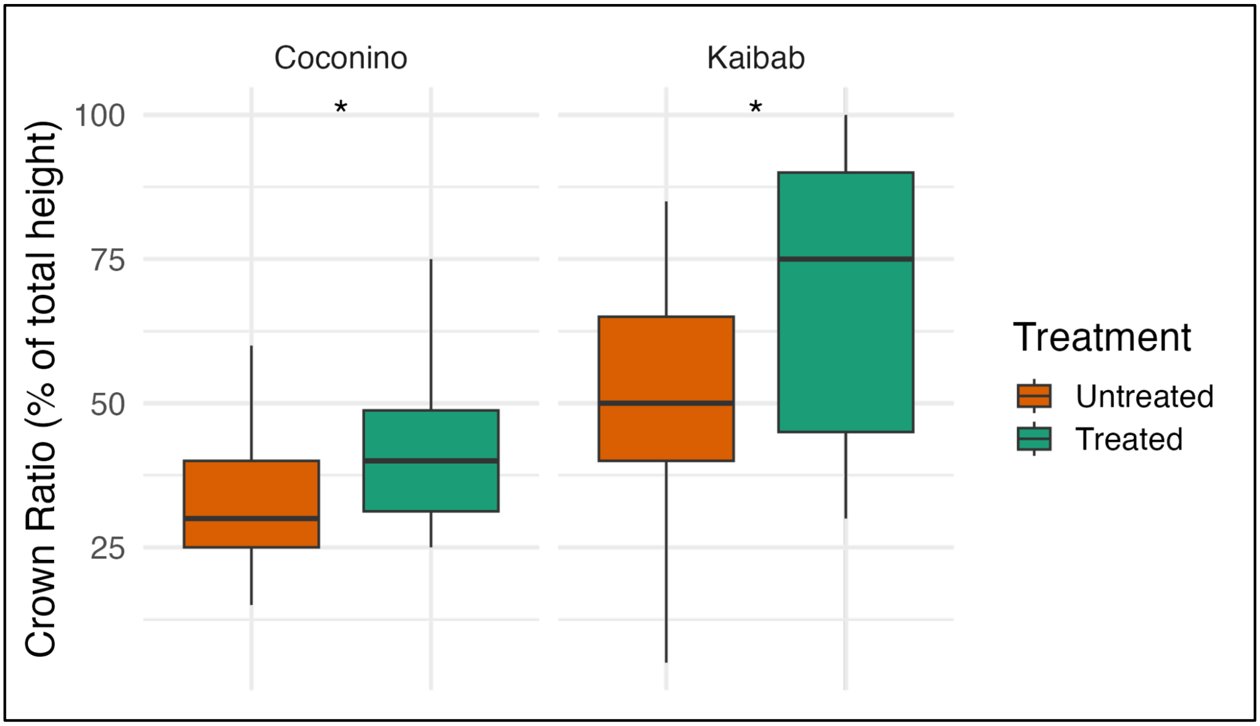

Figure 8. Crown ratio (%) of quaking aspen in treated and untreated units across the Coconino and Kaibab National Forests. Treated units exhibited significantly higher crown ratios than untreated units on both forests (Coconino: p = 0.0398; Kaibab: p = 0.0428; Mann-Whitney U test).

Preliminary Regeneration Response

Regeneration densities (trees per acre; TPA) increased following treatment across both forests and all treatments, with intensity of response varying by treatment type and site (Figure 9). On the Kaibab National Forest, untreated and treated units maintained relatively low regeneration densities prior to treatment (Figure 9). Density then increased immediately in the first post-treatment measurement yet declined by the second post-treatment period (Figure 9).

Treatment effects on regeneration were more pronounced on the Coconino National Forest. Untreated units showed minimal regeneration prior to treatment years and negligible increases over time (Figure 9). Sanitation thinning units exhibited substantial increases in regeneration density observed during the first post-treatment period and maintained through the second annual measurement.

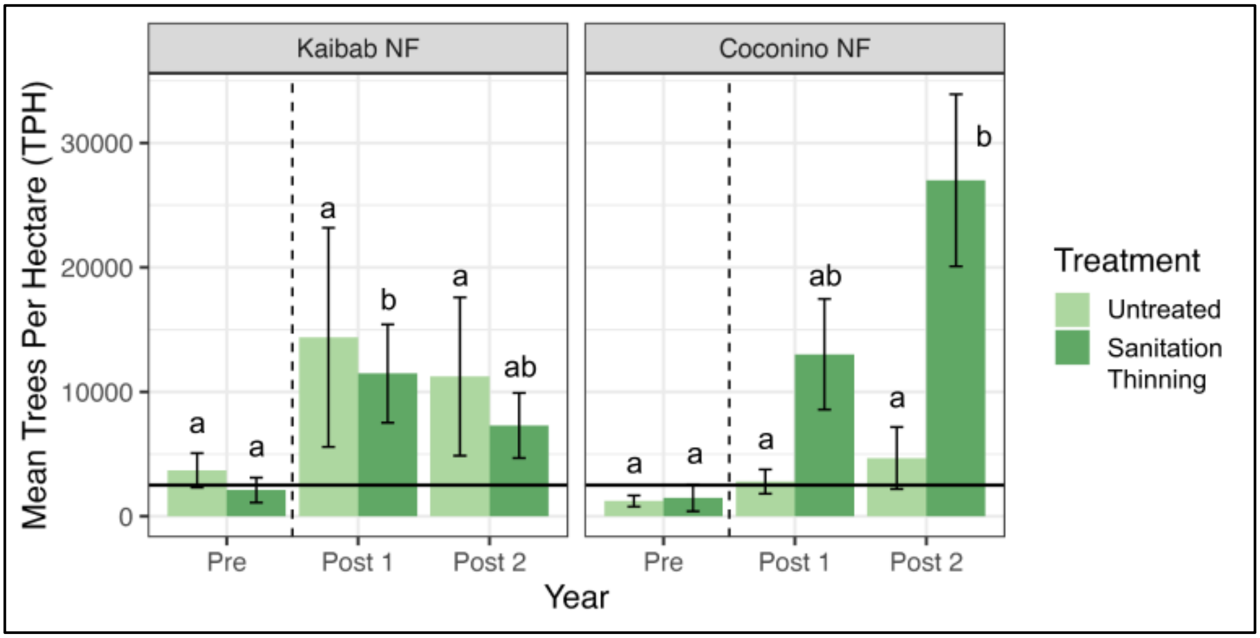

Figure 9. Mean regeneration density size classes (trees per acre; TPH) across pre-treatment (Pre) and post-treatment (Post 1 and Post 2) periods for untreated and sanitation thinning treatments in the Kaibab and Coconino National Forests. The dashed vertical line denotes timing of treatment implementation. Horizontal line denotes 2,500 trees per hectare density threshold defined by O'Brien et al. (2010) for adequate stand self-replacement.

Discussion

Site Comparison

The Coconino and Kaibab National Forests represent contrasting climatic conditions, with the Coconino under cooler and wetter conditions and the Kaibab under warmer and drier conditions (Table 1). In addition, variation in mean tree age and mean residual overstory diameter between the Coconino and Kaibab forests (Table 1) likely influenced pre-treatment baseline conditions and possibly the intensity of a post-treatment response. As a result, treatment responses should be interpreted relative to pre-treatment baseline site conditions when evaluating treatment effectiveness (Ashton & Kelty, 2018, Chapter 3).

The Kaibab site, with a different climate and site history, may have a different response than Coconino sites due to higher pre-existing stress and competition (Carter et al., 2021). Timing of the treatments also varied, with Coconino exclosures being treated in the fall of 2021 and 2022, while the Kaibab exclosure treated in spring of 2020. Site-specific climatic conditions have not yet been researched or statistically analyzed as part of this study; however, similar research has shown a relationship with annual precipitation and warmer peak summer temperatures impacting annual growth (Dudley et al., 2015).

Pretreatment Growth Variability

Variations in stand structure, past disturbance history, and site productivity likely contributed to pre-treatment mean annual BAI differences between forests. Components of stand structure include differences in mean ages between forests, basal area density, quadratic mean diameter (QMD), and relative density (RDI), all of which play important roles in stand-level mean annual BAI. These differences influence resource availability and growth potential, and result in distinct baseline conditions between forests prior to treatment. Additionally, variation in the history of OSS infestations between sites may have contributed to growth differences. Although only one year of OSS data was measured as part of this study, long term impacts from OSS-related mortality populations in the region have been observed previously (Zegler et al., 2012; Crouch et al., 2021, 2024, 2025)

Post-treatment Growth

Treated units displayed higher BAI after treatment compared to untreated units in all forests. An example of an observable BAI growth response following sanitation treatment can be seen in Figure 10. However, the timing of statistically significant responses differed between forests. A one-year delayed response was observed in treated Coconino exclosures, while an immediate response was observed in the treated Kaibab exclosure. This post-treatment lag is likely driven by differences in treatment timing and seasonality (Ashton & Kelty, 2018, Chapter 22). Spring treatments occurred immediately prior to the growing season and allowed residual stems to rapidly utilize newly available resources. In contrast, fall treatments occurred late after the prior growing period and may have delayed measurable growth responses in residual overstory until the following season, constricted by growth seasonal timing and length.

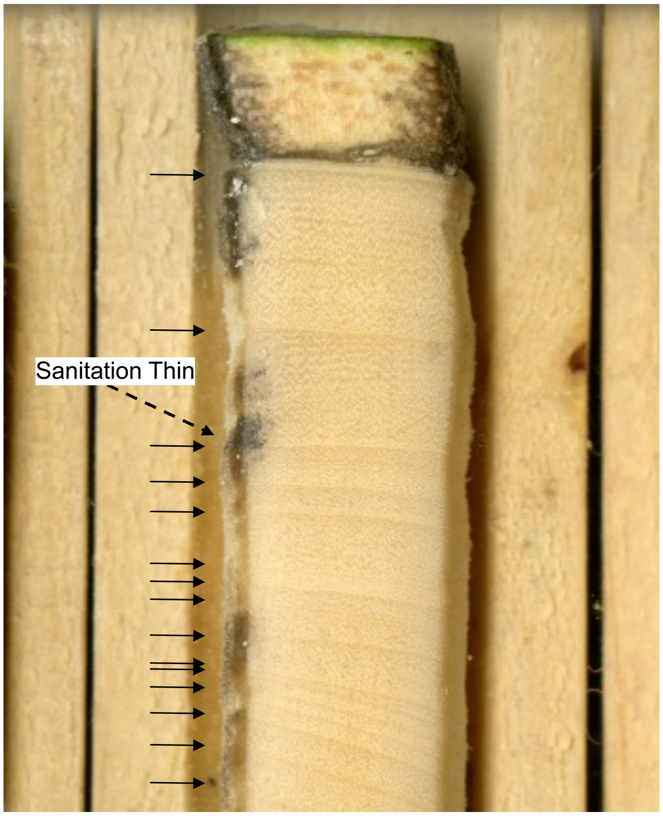

Figure 10. Observable immediate sanitation treatment response in radial growth example from a residual stem on the Coconino National Forest site. Arrows delineate late wood boundaries.

Oystershell Scale Infestation and Tree Growth

Trees with oystershell scale (OSS) observed in 2025 consistently displayed long-term lower mean annual BAI, indicating a history of lower vigor relative to uninfested stems. However, the sustained pattern of reduced long-term growth suggests that these trees were already experiencing chronic stress prior to observed infestation. With a lack of previous data on these specific individual stems prior to being selected for this study, they might have been more susceptible to OSS infestations from stressors, or OSS may also have been directly affecting these stems for years prior to observation. Because OSS presence was only recorded during the 2025 sampling period, the timing of initial OSS colonization cannot be determined.

Periods of reduced BAI aligned with known regional drought stress events of this century (2002, 2018, 2021; National Drought Mitigation Center, 2026), suggesting that climatic stress may have contributed to long-term declines in tree vigor across all stands. Generally, lower BAI is associated with reduced tree vigor and chronic stress, conditions that may lead to increased susceptibility to biotic damaging agents whereas higher radial growth is associated with increased tree vigor (Dobbertin, 2005). This observed pattern supports the interpretation of OSS as a secondary stressor, rather than a primary driver of decline (Manchetti et al., 2011). The current presence of OSS may reflect underlying chronically stressed and less vigorous trees from interacting factors in stand structure and changing climatic conditions (Crouch et al., 2021). Infestation likely occurs preferentially on weakened aspen hosts, with OSS further contributing to decline (Crouch et al., 2021).

Crown Condition

Following sanitation thinning, treated units showed higher mean crown ratios and a lower proportion of severe crown dieback compared to untreated units. These responses may be driven in part by reduced stand density following treatment. Quaking aspen is highly shade-intolerant, and decreased competition can substantially increase light availability within the stand (Ashton & Kelty, 2018, Chapter 22). Increased growing space from greater light availability and reduced competition may have allowed residual trees to capture more available resources, therefore increasing radial growth and expanding crown structure, both through crowns vertically through vertical apical growth, and lateral crown branch extension.

However, the observed improvements in crown condition are also influenced by the prescription itself: the selective removal of infested, poorer-quality stems, which were more likely to exhibit severe dieback or poor crown condition prior to sanitation thinning implementation. This shift is evident at both sites, where prescriptions more likely targeted the removal of smaller trees with a small crown ratio in suppressed and intermediate crown classes. As a result, the remaining population is inherently skewed towards healthier individuals.

Differences in crown condition between treated and untreated units likely reflect a combination of treatment release effects and stand structure shifts resulting from the removal of low-vigor trees. Despite this, the overall pattern of increased crown ratio and reduced dieback suggests that sanitation thinning improved short-term overstory condition and improved long-term resilience to drought and pest disturbances, consistent with silvicultural expectations for this treatment type (Crouch et al., 2023; Chagnon et al., 2025)

Determining Resilience

In forested ecosystems, resilience can be defined as the ability of the stand to recover and maintain its structure and function following disturbance, which depends on both sustained overstory vigor and successful recruitment (Walker et al., 2004; Bryant et al., 2019; Crouch et al., 2025). Both components are necessary for long-term ecosystem persistence, as healthy overstory trees support ecosystem function while regeneration ensures future stand replacement (Crouch et al., 2025).

Aspen’s clonal reproductive strategy makes it a more resilient species, as stems removed by disturbance are replaced with stimulated root suckering, supporting stand renewal (Lieffers et al. 2001). Although the extent of aspen’s sexual reproduction is widely debated (Barnes, 1966; Kreider & Yocom, 2021), it has been observed that seed production and germination rates are limited on drier sites with a mean annual precipitation less than 750 mm year^{-1}, with suckers able to establish and outperform any seedlings from shared root reserves (Kreider & Yocom, 2021). Given the uncertainty of future moisture availability under climate change, establishing new cohorts by stimulating suckers in sites without high disturbance may be a key management strategy to ensure aspen regeneration reliability.

Sanitation thinning itself directly stimulated regeneration. Post-treatment regeneration densities by far exceeded the 2,500 trees per hectare threshold defined by O'Brien et al. (2010) for adequate stand self-replacement (Figure 9), indicating that treated stands are likely capable of sustaining future cohorts. Increased light availability following overstory reduction may further support recruitment and development of this regeneration. Untreated units, by comparison, demonstrated continued stress by lower mean annual BAI, less vigorous crown conditions, and lower regeneration trends over time. Although untreated units in Coconino and Kaibab NFs both saw trends of increased regeneration over the observation period, these units only marginally exceeded the minimum regeneration density threshold.

Individual overstory mortality can act as a natural disturbance (Bartos, 2001), involving the removal of overstory stems which stimulates suckering, mimicking the same process as sanitation thinning. New suckers rely on energy from root reserves to get established (Barnes, 1966; Frey et al., 2003); however, it is thought that southwestern aspen stands have a window for the establishment of suckers prior to energy exhaustion (Rolf, 2001; Shepperd et al., 2015). Consequently, this process may not be sufficient for longer-term stand sustainability under continued and repeated pressures from climate stress, non-native insect and pathogen damage, and ungulate browse.

Results from this study suggest that sanitation thinning improved several indicators of resilience in treated units. Treated units had higher BAI and improved crown condition, both of which are associated with increased tree vigor and greater capacity for resource utilization (Dobbertin, 2005). With root suckering response closely tied to tree vigor and root system health, healthier overstory trees may contribute to a greater capacity for regeneration following future disturbance. Combined, these responses indicate that reducing competition by removal of low-vigor, infested stems can enhance overstory vigor and stimulate regeneration, both of which are essential components for aspen resilience.

Limitations

This study is limited by the relatively short post-treatment observation period. Measurements taken of residual overstory stems within the first few years following treatments captured early responses and radial growth and crown condition, but may not reflect long-term patterns in stand dynamics, regeneration success, and sustained treatment effectiveness. Partial thinning may increase the risk of bole damage, windthrow, and sunscald in residual trees. However, limited variation in treatment prescriptions and site conditions, along with a short post-treatment window, reduce confidence in fully assessing or avoiding these indirect effects.

Difficulty in reliably measuring increment core sample ring widths in aspen has been observed with documented occurrences of false (Jones & Shepperd, 1985) and white rings (Hogg & Schwarz, 1999; DeRose & Gardner, 2010), likely due to severe defoliating events or climatic stress. We did not observe the presence of these atypical false or missing rings in this study; however, their potential presence introduces uncertainty in growth measurements and calendar year assignments.

Additionally, sample size limitations further reduce statistical power, particularly in treated-untreated comparisons and within individual treatment units. This reduces the ability to detect treatment effects and increases uncertainty in estimates of variability. Furthermore, differences in site conditions between forests, including elevation, moisture availability, and mean stand age, introduced variability, which complicates direct comparisons of results.

Implications for Aspen Management

Sanitation thinning treatments appear to be achieving intended short-term outcomes. Observed results of new cohort establishment, increased radial growth, and improved crown condition, indicate that these treatments can effectively enhance overstory vigor and promote regeneration in stands impacted by multiple stressors.

Short-term growth responses may serve as a useful early indicator of treatment effectiveness. This will provide land managers with a measurable marker of stand response before longer-term outcomes, such as reduced OSS infestations severity and extent, improved tree survival, and regeneration success may become evident. Continued monitoring will be necessary to determine whether these early responses contribute to sustained, long-term improvements in stand resilience.

Given ongoing pressures from climate, herbivory, and biotic agents, active management is necessary to sustain aspen stands in the Southwestern United States. As outlined by Crouch et al. (2023, 2025), resilience can be better achieved through management strategies that promote age structure diversification, mitigation of negative impacts from ungulate browse on regeneration, and enhancing management complexity. The results of this study support these recommendations, demonstrating that targeted sanitation thinning can be an effective management strategy for increasing vertical and horizontal structural heterogeneity, improving stand conditions to achieve objectives, and contributing to longer-term resilience.

The most effective strategy to tackle OSS outbreaks and increase aspen resilience will be through an integrated pest management program (Crouch et al., 2021). With continued climatic pressure, sanitation thinning alone does not completely mitigate long-term effects. Therefore, more research is necessary in assessing complementary site-dependent interventions, either at a smaller scale (ungulate protection, prescribed fire, systemic insecticide) or at a landscape scale (biological control agents, assisted migration), to ensure the most effective control (Wittenberg & Cock, 2001), essential for longer-term resilience of aspen.

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