Umatilla National Forest Supervisor's Office
Attn: Blue Mountains Forest Plan Revision
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Submitted via: https://cara.fs2c.usda.gov/Public/CommentInput?project=64157

RE: Comments on Final Assessment Reports for the Blue Mountains Forest Plan Revision Process

Dear Blue Mountains Forest Plan Revision Team: November 7, 2024

Thank you for the opportunity to provide public comment on the Final Assessment reports prepared in support of the Blue Mountains forest plan revision effort. Our organizations engage in collaborative forest restoration with the Forest Service, Tribes, landowners, conservation interests, forest products industries, restoration contractors, elected officials, and many others on the Malheur, Wallowa-Whitman, and Umatilla National Forests. *Attached here are the Zones of Agreement from the Northern Blues Forest Collaborative. (The BMFP's ZOAs were submitted earlier.)*

Below, we identify significant issues of importance to our collaborative partners as well as objectives, desired conditions, and management approaches that help frame those significant issues. We also suggest technical resources that may be suitable for NEPA analysis of significant issues.

I. Forest resilience and fire risk mitigation

Objectives, Desired Conditions, and Management Approaches:

1. Reduce the risk of fast-moving, high severity fire near homes, communities, and critical infrastructure.
2. Accelerate ecological restoration of dry forests as necessary to conserve older trees, restore characteristic old forest conditions, conserve wildlife habitat, and promote forest resilience in the face of climate change.
3. Protect and relink pattern-process feedbacks that restore characteristic dry forest structure, composition, and function.
4. To the extent possible, reduce the extent and spread of invasive species.
5. Mechanically treat an average of 25,000 to 50,000 acres per year per national forest unit over a ten-year period to reduce forest density, manage surface fuels, and shift species composition to species better adapted to future change.
6. Reintroduce fire including prescribed fire and wildfire that reduces surface fuel on 25,000 to 50,000 acres per year per national forest unit over a ten-year period.

Technical resources to assist forest planning:

- Review of relevant peer-reviewed scientific literature, including research conducted in the Blue Mountains demonstrating the need for active management and the efficacy of common forest restoration tools. Found in Appendix 1.

- Eleven years of monitoring data from the Malheur National Forest that characterizes existing fuels and vegetation and changes to fuels and vegetation that results from disturbance, succession, and mechanical treatments. Available upon request from the Blue Mountains Forest Partners.

- Geospatial data of all completed silvicultural operations in the Blue Mountains from 1985 to present. Available upon request from the Blue Mountains Forest Partners.

- Specific recommendations for silviculture and a conceptual framework for silvicultural operations. Found in Appendix 2.

- Near complete LiDAR coverage of the Malheur National showing overstory canopy closure, canopy height, and other forest characteristics that inform restoration effects and restoration needs at a landscape scale. Available upon request from the Blue Mountains Forest Partners.

- Fire severity mosaics at 30m resolution across all national forests in the Blue Mountains from 1985 to 2022 suitable for estimating changes in forest structure resulting from fire. Available upon request from the Blue Mountains Forest Partners.

- Analysis of past management effects on fire spread and fire activity. Analysis ongoing; contact Blue Mountains Forest Partners.

II. Old tree conservation

Objectives, Desired Conditions, and Management Approaches:

1. Protect old trees from uncharacteristic disturbance and use active management to perpetuate characteristic dry old-growth forest conditions over time.
2. Stabilize or increase populations of old trees over the life of the plan.
3. Protect from cutting older trees, defined as trees that were established prior to changes in forest structure, composition, and disturbance regimes in the mid to late 1800s.

Technical resources to assist forest planning:

- Review of relevant peer-reviewed scientific literature, including research conducted in the Blue Mountains describing status and trends of old growth and effects of management regimes on old growth. Found in Appendix 3.

III. Fish and wildlife conservation

Objectives, Desired Conditions, and Management Approaches:

1. Align structure and composition of habitats to fall more closely within a range of variability that increases the probability that native species will persist in warmer drier climates with more fire.
2. Restore unique habitats including but not limited to wetlands, meadows, and hardwood stands so that they are diverse and well distributed across the landscape.
3. Maintain and improve cold-water habitat and enhance hydrologic function of aquatic systems.
4. Analyze wildlife habitat and changes to wildlife habitat via a coarse/fine filter approach.
5. Restore wildlife habitat to maintain viable populations of species within their historical ranges.
6. Maintain or expand the geographic extent of rare trees species such as whitebark pine and western white pine.
7. Restore and increase diverse understory plant communities with an emphasis on flowering plants as drivers of food webs.
8. Increase the footprint and overstory recruitment of aspen ecosystems.
9. Restore meadows to their original soil boundaries and restore hydrological connectivity of meadows.

Technical resources to assist forest planning:

- Literature review of typical silvicultural effects on native biodiversity. Found in Appendix 4.

- A comprehensive list of native terrestrial vertebrate wildlife and their associations with different habitat types. Found at https://b88fa6.p3cdn1.secureserver.net/wp-content/uploads/2023/09/Wildlife-Habitat-Zones-of-Agreement-adopted-May-2023.pdf .

- Description of a filter approach to understanding the likelihood of conservation of terrestrial vertebrates across the Blue Mountains National Forests as management continues to restore forests to be well adapted to future climate and disturbance regimes. This approach is consistent both with adaptation to future change as well as the 2012 NFMA planning rule. Found at https://b88fa6.p3cdn1.secureserver.net/wp-content/uploads/2023/09/Wildlife-Habitat-Zones-of-Agreement-adopted-May-2023.pdf.

- Near complete LiDAR coverage of the Malheur National showing overstory canopy closure, canopy height, and other forest characteristics suitable for analysis of wildlife habitat. Available upon request from the Blue Mountains Forest Partners.

- Fire severity mosaics at 30m resolution across all national forests in the Blue Mountains from 1985 to 2022 suitable for estimating changes in habitat resulting from fire. Available upon request from the Blue Mountains Forest Partners.

- Analysis of LiDAR data, understory vegetation data, and historical snag data to predict marten habitat. Analysis ongoing; contact Blue Mountains Forest Partners.

- Technical review of management of northern goshawks, including information characterizing nest site selection, prey base, and canopy closure at the nest and stand level as well as trends in reproduction and survival in forests managed for old growth versus other management. Available upon request from the Blue Mountains Forest Partners.

IV. Carbon storage

Objectives, Desired Conditions, and Management Approaches:

1. Stabilize forest carbon stocks from catastrophic losses due to uncharacteristic fire.
2. Shift carbon storage in denser forest stands composed of many smaller, drought and fire sensitive trees to stands with fewer, larger, drought and fire-resistant trees.

Technical resources to assist forest planning:

- Review of peer-reviewed scientific literature relevant to carbon management. Found in Appendix 5.

- The Forest Service carbon portal, which provides estimates for total carbon storage, carbon storage change over time, and mechanisms for carbon loss for every national forest in the Pacific Northwest region. This resource should be helpful in establishing a baseline of carbon stocks and projecting potential future changes of carbon stocks as a result of implementing the Forest Plan revision. Found at: https://public.tableau.com/app/profile/usda.forest.service/viz/Carbon_Dashboard_Public_17056983339290/Dashboard

V. Conclusion

Thank you again for the opportunity to provide comments on the final Assessment report. If you have any questions about this letter, please contact Mark Webb, bmfp06@gmail.com

Sincerely,

Mark Webb, Executive Director
Blue Mountains Forest Partners

Katy Nesbitt, Chair
Northern Blues Forest Collaborative

Appendix 1. Review of relevant peer-reviewed scientific literature, including research conducted in the Blue Mountains demonstrating the need for active management and efficacy of common silvicultural operations.

A very large body of peer-reviewed scientific studies have documented significant changes to ponderosa pine and mixed conifer forests in seasonally dry inland ecosystems throughout the American West. Available evidence is unequivocal that low and moderate productivity ponderosa pine and mixed conifer forests have experienced significant change in four respects:

1. Stands historically burned very frequently (i.e., every 5-25 years) but today burn infrequently (i.e., 100+ years without fire) since fire exclusion policies were put into place in the late 1800s.
2. Historically frequent fire was generally low severity fire that burned primarily through grass, litter, and shrubs on the forest floor (surface fire) and resulted in the death of isolated individual overstory trees or small clumps of overstory trees. Contemporary fire perimeters include significant area burned at high severity with very large patches where most overstory trees are killed.
3. Forest stands were historically much less dense, had significantly lower average basal area, had less fuel continuity, more complex horizontal structure, and richer diversity in non-forest habitats.
4. Forest stands were historically composed of a much higher proportion of shade intolerant and fire resistant tree species (e.g., ponderosa pine) and a lower proportion of shade tolerant and less fire resistant tree species (i.e., true fir).

Of particular interest in summarizing differences between historical and contemporary conditions in dry forests of the American west are syntheses or meta-analyses of decades worth of research across a variety of forest types over broad areas. For instance, Falk et al. (2011) summarize a variety of tree ring-based fire histories across the western United States and concludes that frequent surface fire was the norm across seasonally dry forests of the American West. McKinney et al. (2019) synthesized dozens of studies and show that ponderosa pine forests of the Colorado and Wyoming Front Range were historically characterized by relatively frequent fire and low or mixed severity fire effects. Safford and Stephens (2015) synthesized a variety of studies across California and show that mixed conifer forest stands of California were characterized by frequent fire, low stand densities, and were dominated by large, old, fire-resistant tree species (see also Kane et al. 2023 and North et al. 2022).

Reynolds et al. (2013) synthesized information about the American southwest and show that historical stands were characterized by frequent low-severity fire, low forest densities, a mosaic of forest and grassland, and that today's stands are much more vulnerable to high severity fire. Merschel et al. (2021) synthesize available information about historical dynamics in ponderosa pine forests of the Pacific Northwest and conclude that these forests were historically characterized by frequent, low severity fires and are today significantly more vulnerable to stand-replacing fire, drought, and insect disturbance.

Available evidence from across the western US provides somewhat less certainty that there has been significant change over time in the most productive mixed conifer sites composed of lodgepole pine, true fir, and Engelmann spruce. However, a number of studies have documented relatively frequent historical fire and lower historical stand density and average basal area than today in moist stands. For instance, Margolis and Malevich (2016) and Johnson and Margolis (2019) found that current fire free intervals in the wettest parts of northern New Mexico are significantly longer than historical intervals, and high severity fire patches greater than 2.5 acres were historically rare.

A wide range of studies conducted in eastern Oregon find that conditions in dry forests in our region are significantly departed from historical conditions. Heyerdahl et al. (2019, 2002, 2001) completed extensive tree-ring based reconstructions of historical fire and historical forest structure and composition on the Wallowa-Whitman, Umatilla, Malheur, Deschutes, and Ochoco National Forests and found that low severity surface fire that occurred every 10-20 years was typical of a wide range of forest types, including moister and more productive forests. Heyerdahl et al. (2019) found that high severity fire was rare and limited in size across the Deschutes and Ochoco National Forests. Merschel et al. (2018, 2014) reconstructed historical disturbance and successional dynamics across a broad range of forest types on the Ochoco, Deschutes, and Fremont-Winema National Forests and found that low severity surface fire dominated historical fire effects, high severity fire was rare, and forest structure and composition had changed dramatically over the last 150 years. Hagmann et al. (2017, 2014, 2013) used timber inventories completed in the early 1900s to demonstrate that forest stands on the Warm Springs and Klamath Reservations on the east slope of the Cascades were historically very low density stands ranging from 10-25 trees per acre, generally two to ten times less dense than contemporary stands.

Heyerdahl, Merschel, and Hagmann's data collection spanned a broad productivity gradient ranging from xeric pine to moist mixed conifer forests. These studies concluded that all forests ranging from dry ponderosa pine forests to moist mixed conifer forests had significantly lower historical forest densities, lower average stand basal area, and more frequent fire return intervals than contemporary forests. Notably, different authors using very different types of evidence (tree ring evidence in the case of Heyerdahl and Merschel and historical timber inventories in the case of Hagmann) reached very similar conclusions.

The findings of these eastern Oregon studies are corroborated by recent reconstructions of historical forest conditions and fire disturbance dynamics on the Malheur National Forest by Johnston et al. (2018, 2017, 2016). This work demonstrates that xeric pine, dry pine, dry mixed conifer, and moist mixed conifer forest ecosystems on the Malheur NF all experienced relatively frequent (every 8-25 years) fire until fire was excluded from the landscape in the late 1800s. Forest density and average basal area has increased in both dry forests and moist mixed conifer forests over the last 150 years. Results from Johnston et al. align well with estimates of historical forest conditions derived from multiple methods from elsewhere in eastern Oregon. In addition, many of Johnston's dendroecological reconstructions are validated by other methods, for instance analysis of General Land Office (GLO) surveys (Johnston et al. 2018).

Recent research on the Malheur National Forest demonstrates that riparian and special habitats had similar historical fire disturbance regimes as upland forests. Harley et al. (2020) found that most historical (pre 1900) fires that burned upland (more than 300 feet from streams) sites also burned riparian (within 300 feet of streams) sites. Downing et al. (2020) found that a relic yellow cedar grove on the Malheur National Forest in a steep northeast facing drainage at 5,700 feet elevation burned during the same years as dry upland sites during the 1800s and late 1700s.

A small group of researchers have argued that historical disturbance regimes have been mischaracterized and that the extent to which forests have experienced change is exaggerated. Most notably, Baker (2015), Williams and Baker (2012), and Baker (2012) use Government Land Office (GLO) records from the late 19th century to infer historical density, composition, and fire disturbance processes across a number of study areas across the western United States, including a 740,000-acre study area on the east slope of the Cascades and a 990,000-acre study area in the northern Blue Mountains. Williams and Baker claim that less than 40% of the Blue Mountains study area and less than 24% of the east Cascades study area historically consisted of low-density, pine dominated forests that experienced frequent fire.

However, Fulé et al. (2014) and other authors show that diameter classes noted in GLO surveyor notes provide no reasonable basis for inferences about historical fire severity and point out that although the GLO surveyor notes relied on by Williams and Baker frequently report low severity fire, they rarely or never report high severity fire (Stephens et al. 2015, Hagmann et al. 2014). Levine et al. (2019, 2017) found that Baker and Williams's methods overestimated tree densities by 24–667% for contemporary stands with known densities. Baker and William's estimates of historical tree density were double that of estimates Johnston et al. (2018) derived from GLO records on the north end of the Malheur National Forest. Notably, Johnston et al.'s (2018) estimates of historical forest density in the Blue Mountains using GLO records are corroborated by other studies and by tree ring-based reconstructions of historical density whereas Williams and Baker's estimates are not.

An important recent contribution to the scientific literature is Hagmann et al. (2021) entitled "Evidence for widespread changes in the structure, composition, and fire regimes of western North American forests." In this paper, 30 different researchers evaluated hundreds of peer reviewed studies published over the last thirty years and concluded:

> "Based on the strength of evidence, there can be little doubt that the long-term deficit of abundant low- to moderate-severity fire has contributed to modification of seasonally dry forested landscapes across western North America. The magnitude of change in fire regimes and the resultant increases in forest density and fuel connectivity have increased the vulnerability of many contemporary forests to seasonal and episodic increases in drought and fire, exacerbated by rapid climate warming."

Significant shifts in species composition and increases in surface fuels, stand basal area, and forest density have had significant negative consequences for contemporary forest dynamics throughout dry forests of the American West. These consequences are described in detail in a variety of synthesis papers including Bradford and Bell (2017), Millar and Stephenson (2015), Spies et al. (2006), and Hessburg et al. (2005). An important consequence of change over time described by these papers is changes in fire behavior and the effects of fire. The size and extent of wildfires has increased dramatically since the 1980s (Parks and Abatzoglou 2020). The large fires that are increasingly common in the western US are usually associated with very large patches where most trees are killed. One study found that in the last twenty years there has been a four to six-fold increase in the proportion of fires burning at high severity in dry forests of Oregon and Washington relative to historical conditions (Haugo et al. 2019). Other studies show that contemporary fire effects are more severe than fires burning over the last two centuries (Parks et al. 2023, Parks and Abatzoglou 2020). For instance, a study of fire on the pumice plateau region of eastern Oregon found that historical fires were the same size or larger than contemporary fires but that area burned at high severity during historical fires was a fraction of the area burned at high severity by contemporary fires (Hagmann et al. 2019).

Like other regions of the western US, the Blue Mountains are today far more vulnerable to stand replacing disturbance. Recent large fires have left very large (>1,000 acres) patches where all trees were killed. Large stand replacing patches and even-aged stands in the wake of stand replacing fire are characteristic of highly productive forest west of the Cascade crest. However, most forests below 7,000 feet in the Blue Mountains are un-even aged stands that were historically characterized by low severity frequent fire that generally killed individual trees or small (<5 acres) patches of trees. These historical disturbance dynamics facilitated the persistence of old (150-800 year old) shade intolerant trees like ponderosa pine and western larch that are highly resistant to fire, drought, insect, and disease. Large patches where all trees have been killed in contemporary fire perimeters results in even-aged regeneration with little remaining old forest structure and is much less likely to develop old growth conditions over the next 150+ years (Coop et al. 2020, Wright and Agee 2004, Youngblood and Coe 2004, Everett et al. 2000).

Low and moderate severity wildfire can have restorative effects. In particular, wildfire reduces surface fuels which helps reduce the risk of future high severity fire. However, research in dry forests in Washington State shows that wildfire failed to meet many objectives for restoration of forest structure and composition (Churchill et al. 2022). Recent research that evaluates fires in eastern Oregon (including fires in the Blue Mountains) shows that only a small percentage of area burned across a relatively narrow range of typical fire severities resulted in restoration of historical structure (density and average stand basal area), and none of the different fires evaluated restored historical forest species composition (Greenler et al. 2023). Historical fire favored shade intolerant species like ponderosa pine and larch because these species are more fire resistant when young than other species, allowing them to persist and recruit into the overstory. After more than a century of fire exclusion, larger Douglas-fir and grand fir are usually quite resistant to fire and are generally only reliably killed by fire when fire is severe enough to also kill ponderosa pine and larch (Greenler et al., 2023).

A significant consequence of large high severity fires in eastern Oregon is the spread of invasive plant species (Kerns et al. 2020). Ongoing monitoring of recent fire perimeters in the Blue Mountains and peer reviewed studies that include data from ongoing monitoring across the Blue Mountains document extensive invasion of grass species including cheatgrass and *Ventenata dubia* in stands burned at high severity (Prevéy et al. 2024). Stands with invasive grasses are at high risk of future high severity fire that will accelerate the spread of invasive species and retard recovery of native diversity (Pulido-Chavez et al. 2021). There is some evidence from dry forests in California that although low and moderate severity fire can increase diversity and abundance, high severity fire decreases abundance of pollinator species (Tarbill et al. 2023).

There are significant human costs to uncontrolled, high severity wildfire. A study of both dry and mesic forests in British Columbia found that increased fire severity was associated with decreases in culturally significant plants, a decrease in native species richness, and an increase in exotic plants (Dickson-Hoyle et al. 2024). Large fire events are expensive and becoming more expensive. Nationwide Forest Service suppression costs have increased by 630% in the last thirty years. Fire-fighting expenses currently account for between 52 and 55% of the Forest Service's total annual budget and are expected to account for 67% of the agency's annual budget within the next three years (National Interagency Fire Center 2021). Smoke from wildfires has significant negative health effects to communities, including altered immune function, increased susceptibility to respiratory infection, and worsening of asthma, pulmonary disease, and cardiovascular disease (Aguilera et al. 2021, Burke et al. 2021, Reid et al. 2016). Uncontrolled wildfires in the Blue Mountains pose a significant risk to human life and property. The 2015 Canyon Creek Fire destroyed 43 homes in Canyon City, and studies suggest that future large wildfires in the Blue Mountains may pose an even greater risk to communities in Grant and Harney Counties (Ager et al. 2021).

Uncontrolled wildfires with significant area burned at high severity is just one consequence of forest conditions significantly departed from the historical range of variability. The synergistic effects of overstocking in the absence of fire, climate change-driven drought, and insect outbreaks are likely to cause significantly more tree mortality across the American west than wildfire (Reilly and Spies 2016, Littell et al., 2009, Raffa et al., 2008, Williams and Birdsey, 2003). Of particular concern is the loss of older trees, which form the structural backbone of dry forests (Franklin et al. 2013). Old trees are at elevated risk of mortality when young trees compete with old trees for light and water (North et al. 2022, Bradford and Bell 2017, Millar and Stephenson 2015, Fettig et al. 2007, Kolb et al. 2007, Waring and Law 2001, Kolb et al. 1998). Competition is particularly acute when trees are large and young because larger trees have greater leaf area and use more resources (Johnston et al. 2019, Gersonde and O'Hara 2005). As a consequence, older trees are in decline throughout the American West (Lindenmayer et al. 2012, Lutz et al. 2009, van Mantgem et al. 2009). A recently completed inventory of more than 1,500 trees in unmanaged roadless areas on the Malheur National Forest showed that almost a third of trees greater than 150 years of age have died as a result of fire, drought, and insect attack.

The Forest Service's National Insect and Disease Risk Map suggests that, given current mortality trends documented by aerial surveys, the majority Blue Mountains will experience between 16-35% mortality of stand basal area in the next 15 years as a consequence of insect and disease. Other parts of the country have previewed the negative consequences to old-growth trees from the synergistic effects of fire exclusion, increased forest density, drought, and insect attacks. More than 30 million older pines were killed by drought in south central California in just five years between 2011 and 2015 (Asner et al. 2016).

A variety of special habitats are extremely vulnerable to current conditions. For instance, Downing et al. (2020) found that grand fir regeneration is rapidly overtaking yellow cedar regeneration following fire in the Aldrich Mountain botanical special interest area, threatening the persistence of yellow cedar, a species found nowhere else in the Blue Mountains.

Aspen stands in the Blue Mountains provide recreational opportunities and critical habitat for wildlife (Seager et al. 2017, Seager et al. 2013a, Strong et al. 2010, Swanson et al. 2010). Aspen stands are rich in small mammal diversity (Oaten and Larsen 2008) and provide important habitat for elk (*Cervus elaphus*) and deer (*Odocoileus spp.*) (Beck and Peek 2005). Aspen's predisposition to heart-rot creates excellent habitat for primary and secondary cavity nesting species, including birds, squirrels, and mice (Martin et al. 2004, Martin and Eadi 1999, Flack 1976). Over 70 species of diurnal breeding birds were detected in aspen communities in the Blue Mountains (Salabanks 2005). Aspen forests host dynamic food webs that support a diverse guild of raptors and carnivores, including goshawks (*Accipiter gentilis*), bobcats (*Lynx rufus*), bears (*Ursus* spp.), and mountain lion (*Puma concolor*) (Fisher and Wilkinson 2005, Debyle 1985).

Aspen are a relatively short-lived species (up to 120 years) that depends on late season soil moisture and low conifer shading to regenerate by root suckering within stands and around the stands (up to 100-150') allowing stand expansion. Aspen stands can persist for decades without understory regeneration, but aspen stands provide habitat for fewer species without a complex understory and are at-risk of being lost when the overstory becomes decadent after 5-8 decades (Strong et al. 2010, Swanson et al. 2010). Even as aspen provides habitat for a significantly higher bird species richness than the surrounding conifer forests (Dobkin et al. 1995, Turchi et al. 1995), aspen accounts for less than 1% of all forested lands in eastern Oregon, and over 50-80% of aspen cover has been lost (Seager et al. 2013a, Seager 2010, Swanson et al. 2010).

A rare and critically important habitat found in the Blue Mountains are whitebark pine (*Pinus albicaulis*) stands, which are found as isolated groves among subalpine fir forests near tree line. White bark pine is an important contributor to local and regional biodiversity in part because its seeds are large and extremely nutritious. White bark pine is in dramatic decline throughout the United States due to exclusion of low intensity fire, drought, and insect attacks (Goeking and Izlar 2018). There is little information about status and trends in this species in the Blue Mountains, although many stands have burned at highs severity in recent years and many unburned stands are being encroached by fir.

Blue Mountains landscapes are significantly departed from historical conditions and ecosystem functions are currently at significant risk from disturbance and drought stressors. But the real problem that creates a strong need for restoration action is that this situation is likely to become much worse in the future because the climate of eastern Oregon is warming and creating conditions more conducive to drought, insect attack, and high severity wildfire.

Important climate change projections for the Blue Mountains are summarized in Kerns et al. (2018) and more generally for eastern Oregon by Halofsky et al. (2020) and Mote and Salathe (2010). These studies predict:

- A significant increase in summer temperature, a significant decrease in spring snowpack, earlier stream runoff, and more variable precipitation patterns.
- Increasingly large and severe wildfires that involve significant overstory tree mortality. In the aftermath of fire, some areas are expected to transition to different vegetative communities.
- A shift in vegetation communities along elevation and latitude gradients, which may involve replacement of many subalpine and alpine systems with new vegetation communities.

Climate change will result in more drought years and longer and deeper droughts in eastern Oregon than at any other time in hundreds of years. Paleoecology reconstructions suggest that sustained multi-year droughts occurred approximately once every hundred years until the mid 1980s in the Blue Mountains. Between 1990 and 2020, there have been several prolonged drought events (Williams et al. 2020, Mote et al. 2019). Dry forest systems such as those found on the Malheur National Forest are more vulnerable to decreased soil-moisture and will be more prone to forest dieback (Anderegg et al. 2019, Allen et al. 2010). Severe water stress related to more frequent and severe drought will likely lead to accelerated mortality of old trees from insects and disease (Anderegg et al. 2019, Stephenson et al. 2019, Kolb et al. 2016. Cochran 1998). Many large trees will be lost to mortality as these disturbance processes become more extensive in the coming decades (Kerns et al. 2018, Littell et al. 2018, Mote and Salathe 2010).

Directional climate change is expected to have significant negative consequences for special habitats across the Blue Mountains. The increased frequency, duration, and severity of drought has resulted in widespread root mortality and crown loss in mature aspen stands in the Rocky Mountain region. Drought associated with climate change is expected to result in significant new mortality of aspen across its current range, including much of eastern Oregon (Worrall et al. 2013). Climate change has significantly contracted the distribution of whitebark pine at local and regional scales and is associated with increased incidence of bark beetle attacks that have resulted in significant mortality of whitebark (Shepherd et al. 2018, Keane et al. 2017).

Directional climate change (hotter, drier, and longer summers, and decreased snowpack) will intersect with trends in forest successional dynamics associated with the exclusion of fire and other land use changes to create conditions that are even less conducive to safe human communities, the persistence of old-growth trees, and maintenance of native biodiversity. Shade tolerant fir has greater leaf area than shade intolerant ponderosa pine and larch, and transpires more water during photosynthesis, exacerbating drought stress to pine and larch (Johnston et al. 2019, Fettig et al. 2007, Gersonde and O'Hara 2005, Waring et al. 1982). In the absence of fire, ongoing monitoring of Malheur National Forest stands shows that regeneration of shade tolerant fir is outpacing the regeneration of shade intolerant species (Johnston 2017). Shade intolerant species cannot replace the ecological functioning of shade intolerant ponderosa pine and larch. Pine and larch live much longer because their root architecture (their tendency to develop deep tap roots and higher resistance to hydraulic failure) and growth and crown characteristics (thick bark and sparse aerial fuels well off the ground) make them much more drought and fire resistant (Domec et al. 2009). Ponderosa pine and larch also devote greater resources to production of defensive compounds that repel insects and help compartmentalize damage from fire (Smith et al. 2016, McCulloh et al. 2014, Hood and Sala 2015). Grand fir are far more prone to mortality from drought, insects, and root diseases than pine. A number of studies investigating mortality of grand fir in eastern Oregon report 100% mortality of large fir over 10 to 20 years of observations (i.e., Filip et al. 2007, Cochran 1998).

Both peer-reviewed literature about common forest restoration treatments in other part of the country as well as the available evidence from ongoing research and monitoring in the Blue Mountains suggests treatments are achieving important positive ecological results. Of particular interest is evidence that: 1) treatments are moderating fire behavior and mitigating the risk of high severity fire to natural and human communities; 2) treatments are maintaining and enhancing native biodiversity and the structure, composition, and processes that flora and fauna depend on, 3) Blue Mountains landscapes are more resilient and better adapted to future climate and disturbance stressors; and, 4) restoration actions are restoring special habitats.

Hundreds of studies have been published in the last three decades that evaluate the ability of mechanical thinning and prescribed fire to moderate fire behavior and mitigate fire risk. One of the most extensive studies of fuel management was the U.S. National Fire and Fire Surrogate study. The overarching goal of the Fire and Fire Surrogate study was to evaluate the effectiveness and ecological consequences of commonly used fuel reduction treatments (McIver et al. 2013). The Fire and Fire Surrogate study involved a total of twelve treatment sites, seven located in western U.S. states and five located in eastern states. At each site, treatments were designed to thin stands so that 80% of the residual dominant and co-dominant trees would survive a wildfire under 80th-percentile fire weather conditions. Three different treatments—mechanical thinning only, prescribed fire only, and mechanical thinning plus prescribed fire—were replicated within at least three randomly assigned treatment units that measured at least 37 acres in size. A comprehensive summary of the effects treatments across these sites found that the mechanical thinning plus fire treatment was best suited for the creation of stands with fewer and larger trees, reduced surface fuel mass, and greater herbaceous species richness, but that the mechanical thinning plus fire treatment sometimes resulted in invasion of sites by invasive species (Schwilk et al. 2009).

A number of metanalyses and syntheses of fuel reduction projects across the American West, including Davis et al. 2024, Willms et al. (2017), Kalies and Kent (2016), and Martinson and Omi (2013) show that mechanical thinning followed by prescribed fire is generally effective at moderating wildfire severity. The majority of published studies suggest thinning that is not followed by prescribed fire is less effective at moderating fire severity than thinning combined with prescribed fire (e.g., Prichard et al. 2020, Prichard and Kennedy 2014, Schwilk et al. 2009). Some studies suggest that thinning without prescribed fire can increase wildfire severity by adding fine fuels to the forest floor (e.g., Raymond and Peterson 2005). Of particular interest are studies that evaluate the effectiveness of fuel reduction over long time periods. Several recent studies leverage long term data sets to conclude that mechanical thinning treatments such as those being undertaken on the MNF can help moderate fire severity and fire spread and limit crown fire behavior for more than 20 years (Brodie et al. 2024, Hood et al. 2024, Radcliffe et al. 2024).

Two peer-reviewed studies have been published that evaluate the effectiveness of fuel reduction treatments in moderating fire behavior and mitigating fire risk on the Malheur National Forest. One study, reported in Westlind and Kerns (2017), was an experimental design that compared the effects of thinning followed by four different prescribed fire intervals: A five-year burn interval with prescribed fire conducted in the spring, a fifteen-year burn interval with prescribed fire conducted in the spring, a five-year interval with prescribed fire conducted in the fall, and a fifteen-year burn interval with prescribed fire conducted in the fall. All thinning and prescribed fire treatments reduced organic forest floor depth relative to untreated controls. Fall burning was associated with greater overstory tree mortality and an increase of 1,000-hour (≥3" diameter) fuels, but there was little difference in accumulation of smaller diameter fuel associated with frequency or season. All fire treatments reduced conifer regeneration, although fall burning at five-year intervals was most effective at removing conifer regeneration.

A second study, Johnston et al. (2021), evaluated modeled fire behavior both before thinning and for five years after mechanical thinning in the Marshall Devine planning area, one of the first projects completed with CFLRP funding. This study only evaluated the effects of thinning—prescribed fire had not yet occurred in the portions of the Marshall Devine planning area where data were collected. Thinning without prescribed fire significantly reduced modeled crown fire behavior immediately after thinning was completed. Modeled surface fire behavior metrics—flame length, rate of spread, and reaction intensity (the amount of heat energy released by fire)— increased for 1-3 years after thinning was completed. But 4-5 years after thinning was completed, all modeled fire behavior metrics had declined to well below pre-thinning levels, in large part because surface organic layers had been reduced, probably because removal of trees had decreased deposition of needles and increased decomposition (Figure 5.1).

A major goal of silviculture across the Blue Mountains is restoration of forest resiliency at stand and landscape scales. Resiliency refers to the ability of stands to undergo disturbance like drought, wildfire, and insect attack and regain their essential functions (Hollings 1973). Of particular interest is the persistence of old trees, which provide critical habitat functions and form the foundation for stands that are resilient to future change because they have persisted through past climatic and disturbance variability (Marcot et al. 2018, Hessburg et al. 2015, Bull et al. 1997). Increases in stand basal area and forest density have reduced drought resistance of old trees (Voelker et al. 2019). Old trees are at elevated risk of mortality when young trees compete for light and water (Bradford and Bell 2017, Millar and Stephenson 2015, Fettig et al. 2007, Kolb et al. 2007, Waring and Law 2001, Kolb et al. 1998). Competition with grand fir is particularly acute because the greater leaf area of this species uses more water (Johnston et al. 2019, Gersonde and O'Hara 2005).

Restoring historical competition dynamics characterized by low basal area, low stand density, and a relatively higher proportion of shade intolerant species has been shown by a variety of studies to increase the resistance of stands to drought, insects, and fire disturbance effects associated with a warming climate (e.g., Vernon et al. 2023, Tepley and Hood 2020, Vernon et al. 2018, Sohn et al. 2016, Larsson et al. 1983, Mitchell et al. 1983). Tree vigor has been shown to be an important predictor of mortality (Keen et al. 2020, Cailleret et al. 2017, Dobbertin 2005) and fuel treatments have been shown to improve tree growth (Roccaforte et al. 2024, Vernon et al. 2023, Young et al. 2023, Thomas and Waring 2015), increase drought resistance (Vernon et al. 2018), and reduce susceptibility to bark beetle outbreaks (Hood et al. 2016, Zausen et al. 2005). Other tree physiological characteristics, such as resin production, are important chemical defenses against bark beetles (Ferrenberg 2014) and the mobilization of non-structural carbohydrates (NSC) may facilitate growth during periods of stress and recovery following disturbance and seasonal change (Vernon et al. 2023, Tixier et al. 2019, Iwasa and Kubo 1997).

The Southern Blues CFLRP Forest Vegetation and Fuels team has collected data within the Marshall Devine planning area on the south end of the Malheur NF to determine if overstory trees are more vigorous following thinning that frees them of competition. Results demonstrate that trees in thinned stands exhibit greater radial growth and less non-structural carbohydrates in wood fiber, indicating that those elements have been mobilized to produce defensive compounds and leaf, bole, and root mass (Vernon et al. 2023).

The few studies of landscape scale forest restoration demonstrate that forest restoration not only benefits forests within a treated landscape, but can benefit forests outside the treated landscape by modifying landscape patterns of fire and other disturbances (Roccaforte et al. 2024, Remy et al. 2024). One of the few studies to investigate the hydrologic effects of thinning showed that extensive mechanical treatments were associated with increased streamflow and soil water storage (Cederstrom et al. 2024). Finally, a meta-cost-benefit analysis of fuel treatments across the American West showed that every dollar invested in forest restoration results in up to seven dollars returned in benefits (Hjerpe et al. 2024).

Treatments to remove conifers from aspen stands have been shown by previous studies to help mitigate the effects of warming and a decrease in moisture availability associated with climate change. Increasing moisture available to aspen by removing conifers has been shown to support persistence of aspen, aspen growth, and expansion of aspen groves during normal and drought years (Seager 2017, Swanson et al. 2010, Seager et al. 2013a, Seager 2010). Aspen stands where conifers have been removed on the Malheur National Forest and nearby forests show increased resiliency as measured by increase in basal area, stand size, and recruitment of midstory and overstory (Seager 2010). Multi-storied aspen stands with recruiting sprouts were more likely to persist during drought and other disturbances (Worral et al. 2010, Seager 2010).

Appendix 2. Specific recommendations for silviculture and a conceptual framework for silvicultural operations.

The science literature (see Hessburg et al. 2016, Stine et al. 2014, Agee and Skinner 2005, Brown et al. 2004, Franklin et al. 2013, and Franklin and Johnson 2012) provides a number of important principles to guide upland forest restoration treatments in the Blue Mountains, including:

1. Retain all older trees, generally defined as trees that established prior to extensive Euro-American interventions on the landscape beginning in the late 1860s.
2. Improve the survivability of older trees by removing ladder fuels and reducing competition around older trees.
3. Thin forests to reduce forest density and shift composition from late seral shade tolerant species to early seral shade intolerant species.
4. Reduce surface fuels by reintroducing fire to stands following treatment.
5. Increase forest diversity at both the stand and landscape scales by varying treatment intensity, creating openings, and leaving untreated areas.
6. To the extent possible, integrate upland forest restoration treatments with management of invasive species, wildlife habitat, roads, stream crossings, and range developments.
7. To the extent possible, take advantage of opportunities to conduct restoration activities in special habitats like hardwood stands, riparian areas, and meadows.

Upland forest restoration involves three different elements (Churchill et al. 2013):

1. Variable density thinning
2. Openings
3. Untreated areas

Achieving upland forest restoration goals and objectives is a matter of applying these three elements in a spatial pattern appropriate for different stands and landscapes. The application of variable density thinning, openings, and untreated areas should all have specific ecological rationale tailored to site specific conditions.

At a stand scale, upland forest restoration treatments may result in a fine-grained spatial pattern when small-sized openings and untreated areas (0.1 to 0.5 acre) are scattered throughout a matrix of variable density thinning. Treatments may result in a moderately coarse-grained pattern when medium-sized openings and untreated areas (0.5 to 2 acre) are located within a matrix of variable density thinning. In some cases, a coarse-grained pattern may be appropriate in which large areas (2 acres and greater) are left untreated or where all or most trees are removed from a larger area to restore meadow habitat or create conditions for recruitment of species that are very sensitive to conifer competition, e.g., western white pine, western larch (tamarack), and aspen.

The spatial pattern appropriate for stands and landscapes is determined by considering how stands and landscapes will change over time as successional and disturbance processes interact with residual forest structure. As an example, untreated areas may persist as denser, multi-layered stands for many decades if they occupy landscape positions with sufficient water resources and/or if they are relatively insulated from insects and fire within a landscape that has been extensively treated. In other cases, an untreated area may experience stand replacing disturbance within a relatively short period of time and begin functioning as an early seral opening. Openings may persist indefinitely if recurrent disturbance removes trees, or they may quickly regenerate and function as dense forest habitat at some point in the future. All restoration prescriptions should explicitly address how treatments will interact with future vegetation succession, fire, insect activity, climate variability, and future management activities. In particular, restoration prescriptions should be explicitly tied to plans to implement prescribed fire and manage future wildfire. Distances between residual trees and the aggregation of residual forest structure should vary as appropriate given site conditions and objectives. Leaving clumps of trees where older trees or stumps are found in clumps, removing trees from around the canopies of old trees, and removing trees from historically treeless areas all tend to create diverse spatial pattern.

Although the precautionary principle is often interpreted to suggest that managers maintain existing forest structural and compositional elements if there is any doubt as to the effects of active management, this approach to restoration often involves significantly more risk than not taking action. Current federal policy tends to ensure that significant portions of planning areas will not be treated. Although there is a role for untreated areas, in most treatment units an emphasis on openings and variable density thinning with small leave patches and clumps of trees has the highest probability to achieve landscape scale resiliency across the Blue Mountains.

Upland forest restoration treatments should vary across different forest types to reflect different responses of different forest communities to future disturbance processes. Variation in forest types in the Blue Mountains reflects differences in available soil water and atmospheric limits on transpiration (Johnston et al. 2016). Available soil water varies with precipitation, soil depth, and soil type (deep soils and/or soils with significant ash are associated with higher available soil water). Atmospheric limits on transpiration are controlled primarily by vapor pressure deficit, which is strongly correlated with maximum summer temperature (Landsberg and Waring 1997).

It is often useful to consider whether stands have shade tolerant species that were established prior to the early settlement period of the late 1800s when managers and users of the forest began to intentionally exclude fire from the landscape. The establishment of shade tolerant species prior to this period suggests a relatively productive site in which shade tolerant species persisted through drought and fire and can potentially continue to persist in the face of future climatic and disturbance variability. Conservation of older shade tolerant trees like grand fir is important because this species often has complex crowns and is prone to defect and bole cavities, features which are important to a variety of wildlife (Bull et al. 2007, Daw and DeStefano 2001).

The presence of older shade tolerant grand fir is a good way to distinguish between ponderosa pine and mixed conifer forest types. In ponderosa pine stands, 90-100% of basal area of older trees is ponderosa pine. As much as 10% of older trees may be a combination of Douglas-fir and/or western larch (tamarack). There is little or no older grand fir in dry pine stands. Dry pine stands can be further divided into dry and xeric stands. In xeric stands, ≥99% of older basal area is ponderosa pine with scattered older western juniper and mountain mahogany.

Mixed conifer stands have older grand fir. In dry mixed conifer stands, around three-quarters or more of the basal area of older trees is ponderosa pine with the remaining older basal area in grand fir, Douglas-fir, or western larch. In moist mixed conifer stands, less than three-quarters of older tree basal area is ponderosa pine and western larch. Between 10-40% of historical basal area of moist mixed conifer stands is grand fir or Douglas-fir. Western white pine may be present, along with Engelmann spruce and lodgepole pine. Moister mixed conifer stands are often identified by understory species that generally only occur on deep, ashy soils, including twinflower (*Linnaea borealis*), big huckleberry (*Vaccinium membranaceum*), and grouse huckleberry (*Vaccinium scoparium*) (Figure 6.2).

In ponderosa pine stands, average stand basal area should be reduced to between 35 to 60 square feet of basal area. It is most appropriate to meet basal area at the scale of large treatment units, not on a per acre basis, meaning that if the basal area target is 50 square feet, some acres should have 0-10 square feet of basal area while other acres have 90 to 120 square feet of basal area to meet the target. Most if not all shade tolerant trees should be removed from dry ponderosa pine stands, although trees of any species established prior to the late 1860s should be retained. It is not uncommon for a few older Douglas-fir to be encountered in dry pine stands.

Openings play an important role in mediating the behavior of fire and insect disturbance and can be an important source of vegetative diversity. Removing conifers that have encroached into areas that historically had little or no tree cover will often make an important contribution to landscape scale diversity and resilience (Hessburg et al. 2015). Restoring historical openings may involve removing most or all extant forest cover. The restoration of dry pine may result in relatively large openings, but clumps of leave trees should be relatively small (.1-.5 acres) and spatial pattern should be relatively fine grained. The primary opportunity for diversifying spatial pattern in ponderosa pine stands comes from creating openings, leaving isolated older and mature trees as well as clumps of mature and old trees, and leaving small patches of leave trees. While thinning ponderosa pine stands, young trees that will become old growth trees over time should be retained both as scattered individuals and patches or clumps; but the majority of residual basal area should be concentrated in the oldest age classes of ponderosa pine present on the site. Operations in ponderosa pine sites should usually result in a significant increase in mean stand diameter.

In mixed conifer stands, average stand basal area should be reduced to between 40 and 75 square feet of basal area. As with ponderosa pine stands, basal area targets in mixed conifer stands should be met at the scale of large treatment units, not on a per acre basis. Like ponderosa pine stands, we expect that in any given acre of mixed conifer treatments, stand basal area could be from 0-10 square feet of basal area or 100 to 200 square feet of basal area.

Historical successional and disturbance dynamics created somewhat more variable residual tree patterns in mixed conifer stands, and mixed conifer stands typically provide some complex forest habitat. In ponderosa pine stands, an over-riding objective is to ensure the persistence of older ponderosa pine, which is achieved by variable density thinning that reduces forest density and ladder fuels around individual older trees and clumps of older pine and leaving only very small patches of untreated or lightly thinned trees. Only older shade tolerant trees are retained in ponderosa pine stands if present. Protecting older trees is also a goal of treatments in mixed conifer stands, although it is often appropriate to spread residual basal area through a range of size classes, maintain a diversity of species, and leave some complex forest. This will result in an increase in mean stand diameter after treatment, although there is often a smaller post-treatment increase in mean stand diameter than in ponderosa pine stands. "Free selection" may be used in mixed conifer stands to maintain a variety of tree densities, patch sizes, and vertical complexity (Graham et al. 2007). This system can also be used to provide for down wood, snags and decadent older trees. Free selection typically relies heavily on the operators to ensure that desired outcomes as opposed to strict targets are met. Free selection will typically result in highly variable forest stands with small to large openings and small to large leave patches or lightly treated patches. Restoring meadow and savannah habitat is appropriate for mixed conifer stands. Larger openings are also often necessary to provide for the recruitment of western white pine and western larch (tamarack).

An important desired future condition for many forest stands involves widely spaced older early seral species. Age-based rather than size-based cutting limits better achieve resilience objectives. Absent a site-specific analysis that indicates logging older trees is necessary to achieve resilience objectives, trees that were well established prior to extensive Euro-American interventions on the landscape beginning in the 1860s should be protected. Adopting a younger age threshold may be necessary to ensure recruitment of old growth trees when there are few or no older trees present in stands. Leaving sufficient younger trees to perpetuate desired structure and species composition is usually necessary. Protecting trees that exhibit morphological characteristics indicative of old age using existing field guides or new guides under development will help determine which trees to retain during restoration activities (Johnston and Lindsay 2022, Van Pelt 2008).

Traditional forestry practices emphasize leaving healthy and vigorous trees. Younger, vigorous grand fir and Douglas-fir are often the biggest threats to stand resiliency because they compete with older larch and ponderosa pine. The Forest Service should retain late seral species with significant defects which better provide habitat for cavity excavators and other wildlife where appropriate. Older, defective, grand fir in dry and moist mixed conifer sites are excellent wildlife trees.

Spatially extensive treatments are necessary to promote landscape scale resiliency. Restoration treatments should be implemented over as large a scale as possible consistent with economic and planning efficiencies, legal mandates, and other resource management objectives.

Many needed restoration treatments will involve significant investments and will generate few or no receipts. But where possible and consistent with ecological resilience objectives, restoration treatments should be designed to minimize costs while maximizing ecological and economic returns. Environmental analysis should be concise as possible consistent with informing stakeholders and ensuring rigorous compliance with legal obligations.

Aspen stands provide a disproportionate amount of habitat for wildlife on the Malheur National Forest (DeByle 1985, White et al. 1998). Aspen stands that have a complex overstory, midstory, and understory of aspen trees and other shrubs are generally the most productive and support more wildlife and more diverse food webs (Rogers et al. 2014, Seager et al. 2013a, Strong et al. 2010, Swanson et al. 2010, Shepperd et al. 2006,). Stands that are missing one or more of those aspen story components should be prioritized for restoration. The major goal of aspen restoration is to create complex stands that include midstories and/or understories and to expand the spatial extent of stands. These goals are accomplished by stimulating aspen recruitment and protecting young aspen from browsing by ungulates. Aspen can reproduce vegetatively, where buds form on the roots and sprout, forming clonal suckers (or aspen sprouts) that are genetically identical to the parent tree. Aspen can also regenerate by seed. A mix of a variety of different treatments are appropriate to restore aspen, including 1) conifer removal, 2) fencing, 3) raising water tables, and 4) reintroduction of fire.

Because aspen grow on some of the most productive sites in the Blue Mountains (often sites near water or with deep soil), in the absence of fire, aspen stands are highly susceptible to encroachment by conifers that take advantage of high soil moisture and often grow to be quite large in a relatively short amount of time (particularly grand fir). In many cases, fencing is necessary to exclude both domestic and wild ungulates that prefer new aspen suckers and will often overbrowse new aspen, preventing the development of understory canopies and the expansion of aspen stands (Endress et al. 2012). Finally, application of fire is strongly encouraged to restore resilience to aspen stands. Fire removes competing conifer trees, kills mature aspen stems, stimulates root-sprouting, and increases moisture availability within and between aspen stands, which eases herbivory pressure (Seager et al. 2013b, Shinneman et al. 2013). Fire also creates bare mineral soil required for aspen seed to germinate.

Aspen can expand through a sprouting zone that extend 100 to 150 feet from the last mature aspen stem. Aspen can sprout prolifically outside of existing mature stands when moisture and light is available, and conifer removal, fencing, and fire is recommended within existing aspen stands and as far as 150 feet from existing aspen stands (Shepperd 2001). Expanding existing aspen stands makes stands more resistant to drought and herbivory (Seager 2017, Seager et al. 2013b, Seager 2010, Swanson et al. 2010, Keyser et al. 2005).

The persistence of aspen and the response of aspen to treatments can vary dramatically between aspen stands, requiring careful consideration of site-specific conditions while restoring aspen. In some stands, it may be appropriate to remove all conifers within the stand. In very moist portions of aspen stands, conifers may not be competing strongly with aspen and retention of some conifers may increase avian diversity (Griffis-Kyle and Beier 2003). Older ponderosa pine and very widely spaced younger conifers have been shown to have little impact on aspen recruitment. Conifers showing old growth characteristics (Franklin et al. 2013) and conifers with strong potential to replace dead old-growth conifers within aspen stands should be retained in and around the aspen stands (Seager 2017, Seager et al. 2013a, Seager 2010).

In some cases, particularly when aspen stands are in immediate danger of being lost, the best aspen restoration strategy is to reinitiate stands by killing all remaining overstory aspen by prescribed fire, clear-fell coppicing (cutting aspen overstory), or other overstory or root disturbance (Shepperd 2001). Such disturbances greatly increase clonal root-sprouting density and area, allowing the stand to expand.

Chronic herbivory by native and domestic ungulates suppresses aspen suckers, which inhibits recruitment of aspen and development of understory and midstory components of aspen stands that are important to wildlife and stops new cohorts of small diameter aspen trees from recruiting into the overstory (Seager et al. 2013a, White et al. 1998). Fencing and other methods of excluding ungulates from aspen stands (such as jackstrawing felled trees or leaving coarse woody debris) are often appropriate. Limiting herbivory is particularly important following disturbance that removes aspen overstories, which stimulates suckering. Taking a landscape scale view, appropriately timing treatments, and implementing herbivory mitigation measures is critical to the success of aspen restoration. Restoring aspen over a large area will disperse grazing pressure and make herbivory measures easier. Aspen suckers develop into trees with canopies out of reach of ungulates after 10-15 years, generally corresponding to aspen heights of approximately 8 feet tall. One study found that early season use of aspen was less impactful on sucker growth and survival (Jones et al. 2009). Deer generally browse aspen suckers spring through fall. Livestock usually graze grass, forb and shrub understory in aspen stands in the summer and eat aspen suckers in the fall. Elk graze during summer and browse aspen in the fall and winter. Elk can eat many years' worth of growth on an aspen sucker and are usually more impactful than deer. Aspen stands found in winter elk range are at higher risk for chronic browsing. Monitoring browsing of aspen stands is critical to determining which stands being over-browsed. If browsing is suppressing the suckers (50-100% browsed), and none are growing above browse height of 6'-8', then fencing, deterrents, or alternative grazing strategies should be adopted (Seager 2013b, Seager 2010). Beaver may browse aspen and fell overstory trees into perennial stream systems. Flooding that results from beaver dam construction can also enhance aspen habitat.

Whitebark pine (*Pinus albicaulis*) is a five-needled pine that is in steep decline across most of its range because of the combined effect of mountain pine beetle (Dendroctonus ponderosae) outbreaks, fire exclusion, and the spread of *Cronartium ribicola,* an exotic pathogen which causes white pine blister rust and usually kills infected trees. Whitebark pine is a keystone species in subalpine settings where it is found on the Malheur National Forest. This species helps regulate snow melt and reduces soil erosion. Its large and nutritious nut is the foundation for high elevation foodwebs and is important contribution to landscape scale biodiversity (Keane et al. 2012).

Whitebark pine stands in the Blue Mountains are being encroached by true firs in the absence of fire, which makes them more susceptible to mortality from fire and mountain pine beetle. Thinning to reduce fir competition in whitebark pine stands has been shown to increase resistance to insects, disease, and fire and stimulate regeneration (Larson and Kipfmueller, 2012, González-Ochoa et al., 2004, Keane et al. 2001).

Common silvicultural strategies for whitebark pine shown to be effective at increasing resilience of white bark pine stands include thinning of fir and low intensity prescribed fire to release whitebark from competition and stimulate regeneration. The Forest Service should also consider planting of blister rust-resistant seedlings, especially in areas previously occupied by whitebark pine that have been impacted by high severity fire (Maher et al. 2018, Keane et al. 2017). It is often not necessary or desirable to remove all fir from whitebark pine stands. The intent of treatments should be to release immature (non-cone bearing) whitebark pines from competition and create openings sufficient to regenerate whitebark pine and encourage the dominance of whitebark pine.

Riparian systems in a dry forest landscape provide a disproportionate amount of plant and wildlife diversity as well critical ecological services including salmon habitat and drinking water (Naiman et al. 1993, Gregory et al. 1991). Riparian areas across dry forest ecosystems in the West, including the Malheur National Forest, have been significantly degraded by logging, mining, overgrazing, road building, removal of beaver, diversions, and other historical land use activities (Dwire and Kauffman 2003). Of particular concern is conifer encroachment into riparian areas. Conifers, especially shade tolerant species, tend to exclude hardwood trees and shrubs including aspen and willow. Decline of hardwood cover in the Blue Mountains is associated with significant declines species diversity, including bird abundance and diversity (Bryce 2006).

A major goal of many riparian restoration projects should be stabilizing stream banks and restoring native vegetation cover, which often involves removing conifers and planting hardwoods or facilitating the expansion of existing hardwood communities. Increasing moisture availability in the riparian environment by removal of conifers with higher transpiration demands than shrubs is of particular relevance to climate change adaptation (Grant et al. 2013).

Appendix 3. Review of relevant peer-reviewed scientific literature, including research conducted in the Blue Mountains describing status and trends of old growth and effects of management regimes on old growth.

Mortality of trees is increasing across most regions of the western US, with significant consequences for ecosystems and human communities. Of particular concern are declines in the oldest trees within forests (McIntyre et al. 2015, Williams et al 2010, Lutz et al. 2009, van Mantgem et al. 2009). Old trees are associated with unique wildlife habitat, store vast amounts of carbon, provide cool and clean water for aquatic and human communities, and are biological icons with immense spiritual and cultural significance to tribal members and the general public (Marcot et al. 2018, Vosick et al. 2007, Smithwick et al. 2002). Tree mortality is expected to accelerate as the climate warms, and information about status and trends of mortality of different sizes, ages, and species of trees in different landscape settings will inform adaptation to future change (Allen et al. 2015, Cook et al. 2014).

Drought stress and increased evaporative demand associated with rising temperatures are believed to be key drivers of tree mortality in the western US (Moss et al. 2024, Stephenson et al. 2019, Park et al. 2013, McDowell et al. 2008). Drought stress in seasonally dry, fire adapted forests (hereafter, "dry forests") is also likely exacerbated by shifts in forest structure associated with fire exclusion (Hagmann et al. 2021, Stephens et al. 2018). A number of studies indicate that trees in dry forests of uncharacteristically high density are more susceptible to mortality, particularly during droughts (Keen et al. 2020, Restaino et al. 2019, Cailleret et al. 2017, Das et al. 2007). But the relationship between forest structure and mortality may be complicated by the interactions of climate variability and biophysical setting, as well as the presence of insect mortality agents, tree host susceptibility, and other autecological traits (Germain and Lutz 2024, Stephenson et al. 2019, Clyatt et al. 2016, Van Gunst et al. 2016, Hartmann and Messier 2011). Individual tree radial growth is strongly associated with overall tree carbon assimilation and also likely strongly associated with resistance to drought stress and disturbance (Babst et al. 2014, Dobbertin 2005). Studies indicate that reduced radial growth is associated with increased risk of mortality (Germain and Lutz 2024, Roccaforte et al. 2024, Cailleret et al. 2017, McDowell et al. 2008).

An adaptive management strategy for old trees requires information about the status and trends of old trees as well as information about the effects of succession, disturbance, and management on these trends. Several important lessons have already been learned. First, data currently being analyzed for publication demonstrates that old-growth trees that have died over the past decade experienced decreased radial growth prior to death relative to trees that are still alive. Second, recently published research demonstrates that mechanical thinning increases radial growth of residual trees, suggesting that a critical first step in conserving old trees on the MNF is reducing competition from young trees by aggressive thinning (Vernon et al. 2023). Ongoing research and monitoring will generate additional information to be applied in the course of restoration treatments, including high priority areas for old tree conservation and potentially also post-treatment densities appropriate for conserving old trees in different landscape settings.

Appendix 4. Literature review of typical silvicultural effects on native biodiversity.

Earlier research about the influence of thinning and burning on diversity and abundance of plant and animal communities is mixed. A synthesis of results from Fire and Fire Surrogate study sites indicated that plant species richness increased following most thinning and burning treatments (Schwilk et al. 2009). Another synthesis of fuel treatment effects reported inconsistent effects to plant communities from fuel reduction treatments due to the inherent variability in the biophysical environment across the western United States. The most consistent effect of treatments reported in this synthesis was an increase in non-native species (Willms et al. 2017). Yet another meta-analysis of fuel reduction treatments across the western United States showed that total understory plant cover tended to decrease immediately following fuel reduction treatments but tended to increase after approximately 4-5 years following treatment. This synthesis indicated that a combination of thinning and prescribed fire was most strongly associated with invasion of non-native plants, but that non-native plant cover was minimal compared to native cover (Abella and Springer 2015). More recent work seems to indicate a strong positive association between mechanical thinning and an increase in abundance and diversity of native understory vegetation. Springer et al. (2024) found that thinning and prescribed fire nearly doubled native cover and increased species richness by 50% relative to untreated controls across a large network of study sites in the American Southwest. Demarest et al. (2023) showed an increase in native species richness 4-6 years following treatment in the Colorado Front Range.

Two peer reviewed studies describes understory plant response to thinning and burning on the Malheur National Forest. Kerns et al. (2018) report that understory plant cover increased following one application of prescribed fire in a study area on the south part of the forest, but this response was no longer apparent after 10 years. At the end of almost twenty years worth of observations, there was little difference in vegetation cover between unburned sites and sites burned at different intervals. Vernon et al. 2023 evaluated understory vegetation within the Marshall Devine planning area and showed that measures of vegetation diversity increased within several years after thinning (Figure 5.2). Forb cover in particular responds positively to thinning, probably because of an increase in light associated with tree removal, and possibly because the seeds of many forb species (e.g., species of the *Lupinus* genus) germinate following ground disturbance.

In 2018, the Malheur Forest Vegetation and Fuels monitoring team collected pilot data about pollinator diversity in treated and untreated stands in the Marshall Devine planning area on the Malheur National Forest. This data collection is quite limited in scope, but the results were striking. A total of 27 different genera of pollinators in thinned stands were identified versus 12 genera in unthinned stands and 44 unique species in thinned stands versus 24 in unthinned stands. One of the species located in thinned stands was the western bumble bee (*Bombus occidentalis*), which was formerly widespread throughout western North America but whose population has declined dramatically and is now under consideration for listing under the Endangered Species Act (Graves et al. 2020). Although further research will be needed to better understand the effects of thinning on pollinator populations, typical restoration treatments across the Blue Mountains reduce tree cover, increase solar radiation on the forest floor, and probably stimulate flowering plants, all of which are conditions favorable to pollinators (Hanula et al. 2016, Rivers et al. 2018). A larger study of the effects of mechanical treatments on pollinators across the Malheur National Forest is ongoing.

There has been little peer-reviewed empirical research that describes the effects of contemporary restoration treatments on the abundance and diversity of different wildlife species across the Blue Mountains. One meta-analysis of the effects of fuel reduction thinning treatments in the American southwest found that small diameter thinning had slightly positive or no measurable effects on small mammals, rodents, ground foraging birds, passerine bird species, rodents, or aerial-, tree-, or bole-foraging birds (Kalies et al. 2010). Sollmann et al. (2016) found that flying squirrels were found at slightly lower densities in stands where fuel reduction thinning had occurred in the central Sierra Nevadas, but that the overall abundance of flying squirrels within the larger landscape was unchanged. A study of reptiles and amphibians found that repeated thinning and burning treatments that result in decreased canopy cover may benefit lizards but negatively affect salamanders (Matthews et al. 2010). A synthesis of the results of fuel treatments within Fire and Fire Surrogate study sites suggested that most impacts to wildlife were subtle and transient and highly dependent on site-specific variables, and that estimating the effects of restoration treatments on wildlife in the Blue Mountains depends on inherent site variability (McIver et al. 2012).

Appendix 5. Review of peer-reviewed scientific literature relevant to carbon management.

Human civilization and ecosystems face extreme danger from rapidly warming climate caused by anthropogenic greenhouse gas emissions, especially carbon dioxide (CO_2) (IPCC 2018). Forests play an important role in mitigating the effects of climate change because they capture and store CO_2 from the atmosphere (Friedlingstein et al. 2021). More than 90% of carbon stored in terrestrial ecosystems is stored in the world's forests (Pan et al. 2013). Carbon leaves forests and enters the atmosphere via respiration, decomposition, and combustion. But most forests, particularly older forests in the Pacific Northwest, absorb more carbon via photosynthesis than leaves forests via respiration, decomposition, and combustion, resulting in net storage of carbon that helps offset anthropogenic emissions (Hudiburg et al. 2009, see Figure 8.1).

Significant carbon storage in forests is also lost via timber harvest. Timber harvest results in the manufacture of wood products, many of which are designed for long life spans, for instance, dimension lumber that is used in home construction that may last in a home for decades. However, timber harvest results in net carbon emissions for several reasons. First, manufacturing and transporting timber involves significant carbon emissions. Second, a large proportion of timber that is harvested and manufactured becomes manufacturing byproducts, such as sawdust, that becomes atmospheric emissions via combustion or decomposition relatively quickly, even when the end product is relatively long-lived products such as beams or dimension lumber (Hudiburg et al. 2019). Finally, even relatively long-lived wood products that last in a home or other building for decades still typically become atmospheric emissions more quickly than if a tree is not harvested, because unharvested conifers can live for centuries, and persist for many decades as snags or coarse woody debris even after they die (Hudiburg et al. 2009). In short, at stand scales, timber harvest must be viewed as a net carbon emission (Peng et al. 2023, Stenzel et al. 2021, Zhou et al. 2013). Specifically, the carbon emission from timber harvest is equivalent to the carbon emissions involved in transportation and manufacture of wood products, plus the difference between carbon stored in wood products and carbon that would otherwise accumulate in the stand if it were not harvested.

Like all national forests, in the Pacific Northwest, the national forests of the Blue Mountains store marginally more carbon on an annual basis than is lost through decomposition and disturbance. Annual carbon storage in the Blue Mountains is significantly lower than typical national forests in the Pacific Northwest, because forests across the Blue Mountains are relatively less productive than other forests in the region, particularly highly productive coastal Douglas-fir dominated stands in western Oregon and western Washington (McKinley et al. 2022). The primary sources of carbon emissions in the Blue Mountains besides background respiration and decomposition inherent to all forests are insect mortality (which transfers carbon from live pools to dead pools where they decompose more rapidly than live pools), wildfire (which results in combustion of small amounts of carbon and also results in transfer from live to dead carbon pools), and timber harvest (which, as discussed above involves significant carbon emissions and transfers carbon from live tree pools to wood products pools that are released to the atmosphere more rapidly).

Actions to manage these different sources of carbon loss may involve carbon storage tradeoffs. For instance, losses of carbon associated with insect mortality may influence subsequent fire behavior at different time scales. Conversely, fire may increase or decrease susceptibility of forests to insect mortality at different time scales (Carter et al. 2022, Fettig et al. 2022). Thinning harvests involve carbon losses, but may also reduce the extent of high severity fire. There has been no empirical research that quantifies the effects of different active management strategies on carbon stocks across the Blue Mountains, and outcomes of different disturbances and active management strategies may have highly variable effects on carbon stores (Restaino and Peterson 2013). But deepening drought and increasing fire extent and severity throughout eastern Oregon (Parks and Abatzgolu 2020) suggests that much of the carbon currently stored in the Blue Mountains is increasingly vulnerable to loss over the next several decades if stand densities remain at their current levels (Stephens et al. 2020, Halofsky et al. 2020, Kerns et al. 2018). Empirical research in similar seasonally dry forests suggests that these forests are currently storing more aboveground tree carbon than existed historically, and that thinning and reintroduction of fire can help stabilize carbon stocks over long time frames, especially as the climate warms (Young and Ager 2024, Foster et al. 2020, Stephens et al. 2020, Hurteau et al. 2019, Krofcheck et al. 2019, Liang et al. 2018, Hurteau et al. 2016).

As noted above, at a stand scale, timber harvest always results in carbon losses relative to a no-harvest alternative. However, both national and global use of wood products continues to rise as a result of increasing demand for housing and urbanization (Johnston et al. 2023, Peng et al. 2023, World Bank 2022). And although wood products manufacturing involves significant carbon costs, the costs of replacement material (steel, brick, etc.) are even higher. As a consequence, foregoing timber harvest in the Blue Mountains does not mean that there is less CO_2 entering the atmosphere. Given increased demand for wood and in the absence of federal legislation or international treaties that restrict carbon emissions, when timber harvest planned for the Blue Mountains does not occur, equivalent timber harvest that would otherwise have not occurred may simply occur in a different location (Gren et al. 2016). Alternatively, wood harvested from the Blue Mountains may be replaced by material with even larger carbon emission footprints (Bergman et al. 2014).

Put yet another way, although it is possible to quantify decreases in potential carbon storage from timber harvest across the Blue Mountains, it is likely to be difficult if not impossible to demonstrate that foregoing timber harvest at the stand or project scale results in decreased atmospheric CO_2. It may be better to focus on the multiple co-benefits of thinning practices, including fire risk management, improved wildlife habitat, enhancement to stream and watershed health, etc. (Hessburg et al. 2021, Johnston et al. 2021b, Fontaine and Kennedy 2012, Lehmkuhl et al. 2007). The Forest Service can continue to track carbon storage on across the Blue Mountains using existing online tools. The Forest Service should continue to plan and implement extensive forest restoration projects on the assumption that these projects will result in greater net long term carbon storage and more stable carbon storage, while achieving multiple co-benefits for ecosystems and human communities (Anderegg et al. 2022, Bernal et al. 2022,).

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