# Integrated Upland Zones of Agreement

**Version 3.0**
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**by the Blue Mountains Forest Partners**

# Table of Contents

# Introduction

## 1. The Blue Mountains Forest Partners

The Blue Mountains Forest Partners (BMFP), established in 2006, is a diverse group of stakeholders who work together to create and implement a shared vision to improve the resilience and well-being of forests and communities in the Blue Mountains. The work of the BMFP takes place on the 1.7 million-acre Malheur National Forest located in Grant, Harney, and Baker counties in eastern Oregon. The Malheur National Forest is one of several dozen high priority landscapes that receive funding under the Collaborative Forest Landscape Restoration Program (CFLRP, Public Law 111-11) to accomplish accelerated restoration to restore forest resiliency (Schultz et al. 2012). The CFLRP explicitly encourages collaborative, science-based restoration and the Malheur National Forest has some of the most ambitious forest restoration targets of any national forest in the country.

These Integrated Zones of Agreement (Integrated ZOA, or IZOA) are designed to help guide planning and implementation of restoration actions across a broad range of upland forest types, riparian areas, and special habitats found within the Malheur National Forest. The zones draw on an intensive collaborative process that included dozens of meetings and field trips with scientists, managers, local residents, and representatives from the timber industry and conservation groups. These IZOAs incorporate lessons learned from restoration projects that implemented guidance found in previous zones of agreement documents. The IZOA are intended to be the cornerstone of an adaptive management strategy. Ongoing monitoring and research programs are investigating a variety of questions related to forest response to restoration treatments. Results of monitoring and research will be shared with the Blue Mountains Forest Partners and the Malheur National Forest, and these zones of agreement will be updated regularly to reflect lessons learned.

The second section of these zones of agreement describes the BMFP's goals and objectives for restoration and how we weight evidence that informs our suggestions for restoration. The third section describes variation in different vegetation communities across the Malheur National Forest—restoration treatments must be tailored to different vegetation types. The fourth section synthesizes available science that points to the need for restoration treatments, with a particular emphasis on research conducted on the Malheur National Forest to characterize departures from historical conditions and sustainable future conditions. The fifth section synthesizes what is known about the effects of restoration treatments similar to those proposed in these zones of agreement, with a particular emphasis on monitoring conducted on the Malheur National Forest of past restoration treatments. The heart of this zones of agreement document is the sixth section, where we describe process-based restoration treatments. The last three sections provide suggestions for fire management, wildlife management, and management of carbon stocks. We conclude by describing current and planned monitoring, research, and adaptive management.

These IZOA represent non-binding suggestions from the BMFP to the Forest Service. This guidance is meant to be flexible and subject to modification by the Forest Service in response to site specific conditions, new information, public comment, and emerging management challenges and opportunities.

# Integrated Upland Zones of Agreement

## 2. Integrated Zones of Agreement - Goals, Objectives, and Methods

### 2.1 Goal and objectives of restoration

The over-arching goal of our Integrated Zones of Agreement is to serve the mission of the Blue Mountains Forest Partners, which is to:

> Support steady progress towards the long-term goal of the Malheur National Forest as a healthy, diverse ecosystem that is resilient to natural and human disturbance, contributes economic value to area communities, and helps ensure our communities are safe from wildfires. Project selection and scale of execution is such that major restoration at the forest-wide scale will be evident within 30 years.

**Several principles inform our work towards this goal:**

1. We focus simultaneously on the social, economic, and ecological dimensions of management.
2. We focus on multiple scales from forest stands to landscapes, from individuals to communities, and from stream reaches to river basins.
3. We work with dynamic human and natural systems and anticipate significant changes to these systems over time as a result of natural and anthropogenic disturbance.
4. We make use of the best available science and strive to constantly learn and adapt management to new information and ecosystem change.
5. We act strategically given limited time and resources to effect meaningful ecological change.

**BMFP has adopted the following objectives to measure progress towards our goal:**

1. To the extent possible, integrate multiple restoration opportunities in project planning and implementation, i.e., upland forest restoration, stream restoration, road management, fire management, range improvements, etc.
2. Produce and support diverse goods, services, and employment opportunities from national forest lands, i.e., wood products, biomass, recreation, etc.
3. Plan and implement projects efficiently, effectively, and with maximum impact in compliance with applicable law and regulation.
4. Reduce the risk of fast-moving, high severity fire near homes, communities, and critical infrastructure.
5. Protect old-growth trees from uncharacteristic disturbance and create and perpetuate appropriate old-growth forest structure over time.
6. Restore unique habitats including but not limited to wetlands, meadows, and hardwood stands so that they are diverse and well distributed across the landscape.
7. Maintain and improve cold-water habitat and enhance hydrologic function of aquatic systems.
8. Create conditions for healthy populations of native plants and animals.

9. To the extent possible, reduce or eliminate the extent and spread of non-native invasive species.
10. Manage roads and other infrastructure that serves human needs and is appropriate for accomplishing ecological and wildlife objectives.

### 2.2. Methodology for achieving objectives

We offer suggestions for management and help facilitate planning and implementation in support of those suggestions.

Our suggestions focus on active management. Decisions to take action and decisions not to take action or defer action have negative and positive consequences to resources at different spatial and temporal scales. Our suggestions are based on the best available evidence about whether action is needed and what action is needed to accomplish the objectives above at different scales.

**Evidence that we use to make suggestions includes:**

1. Peer reviewed scientific papers that address management actions on the Malheur National Forest, including action suggested by the BMFP.
2. Peer reviewed scientific papers that address management actions that are similar to those suggested by the BMFP or within similar regions or ecosystems.
3. Peer reviewed science syntheses that address management actions similar to those suggested by the BMFP.
4. Reports and presentations that describe ongoing multi-party monitoring programs on the Malheur National Forest.
5. Reports and presentations from BMFP members, external partners, scientists, and experts that present information, technical data, analysis, or data synthesis from ongoing research on the Malheur National Forest or from similar regions or ecosystems.
6. The experience, professional judgement, and expertise of our members, external partners, scientists, and experts that are synthesized in the course of meetings, field trips, and a variety of formal and informal interactions facilitated by the BMFP.

Some types of evidence we use to make suggestions are inherently more rigorous and reliable, but all of the above types of evidence are salient and useful. We never have access to perfect or unequivocal evidence. We acknowledge that many of our suggestions for active management involve uncertainty and/or risk of negative consequences to resources. We suggest active management only when carefully weighing the available evidence strongly suggests that passive management or deferring active management to a later date involves greater uncertainty and/or risk. Monitoring and adaptive management are essential to ensuring that we understand management tradeoffs and make informed management choices.

## 3. Vegetation types

### 3.1 Variability in forest vegetation on the MNF

The key to effective landscape-scale restoration is accounting for variability across sites and accurately characterizing 1) vegetation diversity, 2) inherent site productivity, and 3) past, present, and future disturbance and successional dynamics.

Common conifer tree species on the MNF include ponderosa pine (*Pinus ponderosa*), western larch (*Larix occidentalis*), Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), lodgepole pine (*Pinus contorta*), and western juniper (*Juniperus occidentalis*). These species can be roughly divided into two different types of tree species: 1) early seral, shade intolerant trees (e.g., ponderosa pine and larch); and, 2) late or mid seral, more shade tolerant species (e.g., grand fir and Douglas fir).

Some species do not fit easily into these categories. For instance, lodgepole pine is relatively shade intolerant and an early seral species in cooler environmental settings but late seral in warmer upland mixed conifer stands. Western juniper is capable of establishing under the canopy of taller conifers, although mature juniper is shade intolerant.

Less common MNF tree species that are found within a narrower range of habitats include subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), western white pine (*Pinus monticola*), and whitebark pine (*Pinus albicaulis*). Subalpine fir and Engelmann spruce are relatively rare at low and mid elevations on the MNF but will be encountered during restoration treatments in moister stands. Western white pine and whitebark pine are in decline across the west and are found in a few isolated locations on the MNF that should be a priority for restoration.

An important consideration when planning restoration that adapts the landscape to future change is the relative drought tolerance of different tree species. There is a strong negative relationship between shade tolerance and drought tolerance. In other words, shade intolerant species – ponderosa pine and western larch - are generally much better adapted to drought (Niinemets and Valladares 2006). Shade intolerant species are also generally more resistant to fire. Table 2.1 shows the relative fire and drought tolerance of different MNF species. The data presented in this table is based on peer reviewed literature. Ongoing research and monitoring data from the MNF provides a more nuanced understanding of different tolerances of different species. For instance, unpublished monitoring data shows that grand fir and Douglas-fir are significantly more vulnerable to fire than ponderosa pine when they are very young. But mature and old Douglas-fir are highly resistant to fire and large grand fir may be nearly as resistant to fire as Douglas-fir of the same size. Although inexact, the rankings presented in Table 2.1 confirm there are important differences in drought tolerance between shade tolerant and shade intolerant species. The importance of these distinctions is discussed further in the Need for Restoration and Upland Silviculture Prescriptions sections.

| | Shade tolerance | Drought tolerance | Fire tolerance |
|---|---|---|---|
| Western larch | 0.27 | 0.48 | 0.61 |
| Ponderosa pine | 0.32 | 0.86 | 0.77 |
| Western juniper | 0.33 | 1.00 | 0.23 |
| Lodgepole pine | 0.35 | 0.80 | 0.39 |
| Douglas-fir | 0.56 | 0.52 | 0.49 |
| Western white pine | 0.59 | 0.48 | 0.60 |
| Grand fir | 0.8 | 0.46 | 0.42 |
| Engelmann spruce | 0.91 | 0.52 | 0.26 |
| Subalpine fir | 0.97 | 0.40 | 0.31 |

***Table 3.1.*** *Relative drought and fire tolerance of selected tree species found on the MNF. Species are ranked on a scale of 0 to 1 with 1 being the greatest relative tolerance. Species are listed from least to most shade tolerant. Shade and drought tolerance data is from Niinemets and Valladares (2006). Fire tolerance data is from Stevens et al. (2020).*

## 3.2 Hardwoods and other vegetation communities

Quaking aspen (*Populus tremuloides*) is an important hardwood species on the Malheur because of its remarkable value as wildlife habitat as well as human appreciation for the cool shady refugia it provides. Aspen is found in pure stands or in mixed stands with conifer trees. As described in Section 6.4, aspen should be an important priority for restoration.

Mountain mahogany (*Cercocarpus ledifolius*) is a tall shrub or short-statured hardwood tree found on arid sites on the MNF. It is drought tolerant, somewhat shade tolerant, and highly sensitive to fire. Retaining mountain mahogany in stands is often a goal of management because it is an important big game browse species.

There are huge variety of important woodland (defined as canopy cover <10%) or non-forest sage steppe vegetation communities on the MNF. Many of these communities require restoration. Common restoration actions in woodlands and non-forest sage steppe lands involves treatments to remove invasive annual grasses like cheatgrass and removal of juniper and/or shade tolerant fir which have encroached onto sage-steppe lands in the absence of fire.

Herb and forb communities associated with riparian habitats should be a priority for restoration. Typical restoration actions include removal of conifers that have shaded out native grasses and flowers in meadows, wetlands, and riparian areas. Of particular interest to the BMFP are actions to restore willows (e.g., *Salix amygdalloides, S. bebbiana, S. commutate, S. exigua, S. geyeriana, S. lasiandra, S. lasiolepis, S. lemmonii, S. rigida, S. scouleriana*) and other riparian hardwoods including black cottonwood (*Populus trichocarpa*), alder (*Alnus* spp), and aspen. We describe restoration opportunities and priorities in these vegetation communities in Section 6.6

There is currently little scientific evidence that vegetation dynamics in high elevation subalpine fir forests are significantly departed from the natural or historical range of variability and there is no consensus among BMFP members that these forests are in need of restoration. An important exception are white bark pine stands which are found among subalpine fir forests near tree line. White bark pine is an important contributor to local and regional biodiversity and is in decline throughout the United States (Goeking and Izlar 2018). The BMFP supports active management that helps adapt whitebark pine stands to future change by removing competing true fir and/or introducing low severity fire (Maher et al. 2018).

## 3.3 Vegetation classification systems

There are a number of different ways to classify forest vegetation on the MNF. The most common vegetation system used by Forest Service managers is a potential natural vegetation hierarchical system that divides forest vegetation into increasingly broad groupings. The finest scale classification of the potential natural vegetation hierarchy are potential vegetation types (PVTs), which take their name from the dominant overstory tree and understory vegetation assemblages that would develop in the absence of disturbance. Examples of common PVTs across the Malheur NF include 1) grand fir/twinflower, 2) grand fir/elk sedge, 3) ponderosa pine/common snowberry, 4) ponderosa pine/pinegrass, and 5) ponderosa pine/bluebunch wheatgrass.

PVTs are aggregated into Plant Association Groups (PAGs), which are cross-combinations of four different temperature classes (cold, cool, warm, and hot) and four different moisture classes (wet, very moist, moist, and dry). There are a total of 13 PAGs that potentially occur on the Malheur NF (cold very moist, cold moist, cold dry, cool wet, cool very moist, cool moist, cool dry, warm very moist, warm moist, warm dry, hot very moist, hot moist, and hot dry). These PAGs, representing distinctive temperature and moisture regimes, occur across seven different physiognomic classes (upland forest, upland woodland, upland shrub, upland herb, riparian forest, riparian shrub, and riparian herb). Combinations of PAGs and physiognomic classes are organized into the broadest classification—Potential Vegetation Groups (PVGs).

The majority of the Malheur National Forest and the majority of forest restoration activities take place in warm-dry PAGs and dry upland forest PVGs. BMFP also supports restoration activities in hot-dry PAGs (also in the dry upland forest PVG) and in cool-moist PAGs (moist upland forest PVG). For example:

- Grand fir/twinflower is a PVT found within the cool-moist PAG (moist upland PVG) where the BMFP supports mechanical thinning and fire to restore forests.
- Grand fir/elk sedge, ponderosa pine/snowberry, and ponderosa pine/pinegrass are all very common PVGs found in the warm-dry PAG (dry upland PVG) where significant restoration activities have occurred over the last 10 years and where we anticipate significant additional restoration will occur over the next decade.

- Ponderosa pine/bluebunch wheatgrass is a PVT found within the hot-dry PAG (dry upland forest PVG) where restoration is often appropriate.

More details about the PVT–PAG–PVG hierarchical vegetation classifications system are found in Powell et al. (2007).

A number of other forest classification systems have been mapped or otherwise described in the scientific literature and in earlier versions of BMFP's upland forest restoration zones of agreement. A variety of literature (e.g., Johnston 2017, Johnston et al. 2016, Merschel et al 2014) distinguish between:

- "Ponderosa pine" forests in which the vast majority (generally more than 95% of historical and contemporary basal area) is composed of ponderosa pine; and,
- "Mixed conifer" forests which historically had a significant proportion of species other than ponderosa pine (generally more than 10%) and which today have a significant proportion of total basal area composed of grand fir.

Johnston et al. 2016 further divides ponderosa pine and mixed conifer forests into:

- Xeric ponderosa pine: Dominated by ponderosa pine but on hot dry sites or sites with skeletal soil that usually have significant infill of juniper. Understory vegetation often includes sage and perennial bunchgrass.
- Dry ponderosa pine: Dominated by ponderosa pine with soil moisture sufficient to permit some infill of Douglas-fir or grand fir.
- Dry mixed conifer (grand fir dry): A mix of ponderosa pine, grand fir, Douglas-fir, and often larch.
- Moist mixed conifer (grand fir moist): A mix of species, but with ponderosa pine generally a minor species or absent and with significant older grand fir (grand fir established before the 1870s).

Powell et al.'s hierarchical classification system focuses on characteristic overstory tree and understory vegetation species composition as a surrogate for temperature and moisture regimes. Other forest classification systems focus on different current or historical forest structural and compositional configurations. There is considerable overlap between these systems, but they do not crosswalk perfectly (Figure 3.1).

Assessing current vegetation and the inherent productivity of a site is necessary for planning restoration treatments. But the most important consideration is an assessment of the most likely future successional and disturbance processes. When restoring upland forests, for example, it is important to determine whether the site has grand fir that were established prior to fire exclusion policies, and if so, what extent of grand fir cover is likely to be sustainable in the years to come given future climate and disturbance regimes. One helpful way to distinguish between upland forests is simply between "persistent grand fir" stands where grand fir has occupied the site for hundreds of years and all other stands where grand fir was historically absent (Johnston 2017, Johnston et al. 2016, Merschel et al. 2014) (Figure 3.1).

The following page displays Figure 3.1, a chart showing how all these classifications overlap.

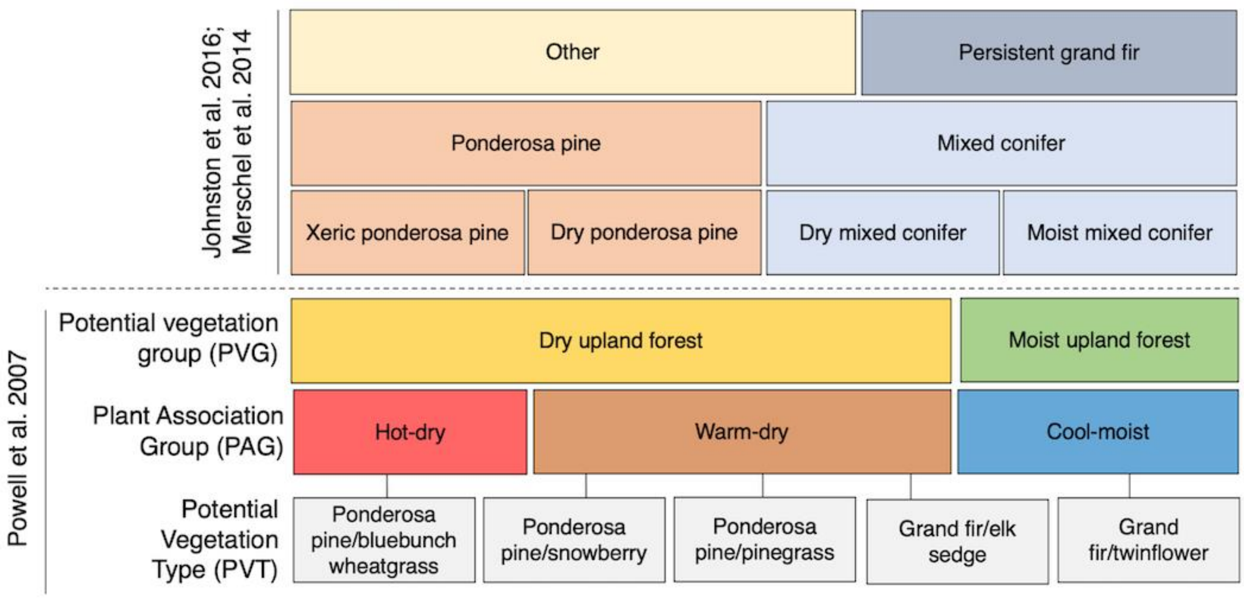



*__Figure 3.1__. Top panels: Vegetation typologies presented by Johnston et al. (2016) and Merschel et al. (2014). Forests across the MNF could be divided into "persistent grand fir" and all other forest types, or, alternately, "ponderosa pine" and "mixed conifer," which can be further divided into "xeric" and "dry" ponderosa pine and "dry" and "moist" mixed conifer. These typologies overlap somewhat with Forest Service vegetation hierarchies presented in Powell et al. (2007) in common usage. The bottom panels show two common PVGs, three common PAGs, and five of the most common of hundreds of PVTs. Note the overlap with Johnston and Merschel typologies.*

## 4. The need for restoration

Restoration actions are appropriate when they are needed in order to achieve the BMFP's goals and objectives (see Section 2.1). Restoration actions are not appropriate when the BMFP's goals and objectives can be served through passive management or by deferring active management. Active management often involves tradeoffs in our ability to achieve the BMFP's objectives at multiple spatial and temporal scales. For instance, active management may involve short term increases in fire hazard followed by long term reductions in fire hazard, or short-term impacts to sensitive species followed by long-term benefits to those species. Careful planning is necessary to ensure that adverse effects to resources do not occur over large areas or over long time periods. The negative and positive consequences of choosing active management over a particular area over a particular time period should be carefully weighed against the negative and positive consequences of choosing passive management.

We evaluate the need for restoration based on the best available information (see Section 2.2), including peer and non-peer reviewed science as well as the practical on-the-ground experience of managers and stakeholders. There is significant evidence that much of the Malheur National Forest landscape is in need of active management, including but not limited to mechanical thinning, re-introduction of fire, and restoration treatments of roads, streams, and degraded special habitats. The need for restoration falls under three major themes: 1) MNF landscapes are significantly departed from historical conditions; 2) MNF forest stands are currently highly vulnerable to uncharacteristic disturbance dynamics that have significant negative consequences to human and natural communities; and, 3) MNF landscapes are poorly adapted to future conditions.

## 4.1 Historical conditions

**Science syntheses about change over time across the western US:**

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:

- Stands historically burned very frequently (i.e., every 2-20 years) but today burn infrequently (i.e., 100+ years without fire) since fire exclusion policies were put into place in the late 1800s.
- 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.
- 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.
- 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 Stevens (2017) 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. 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

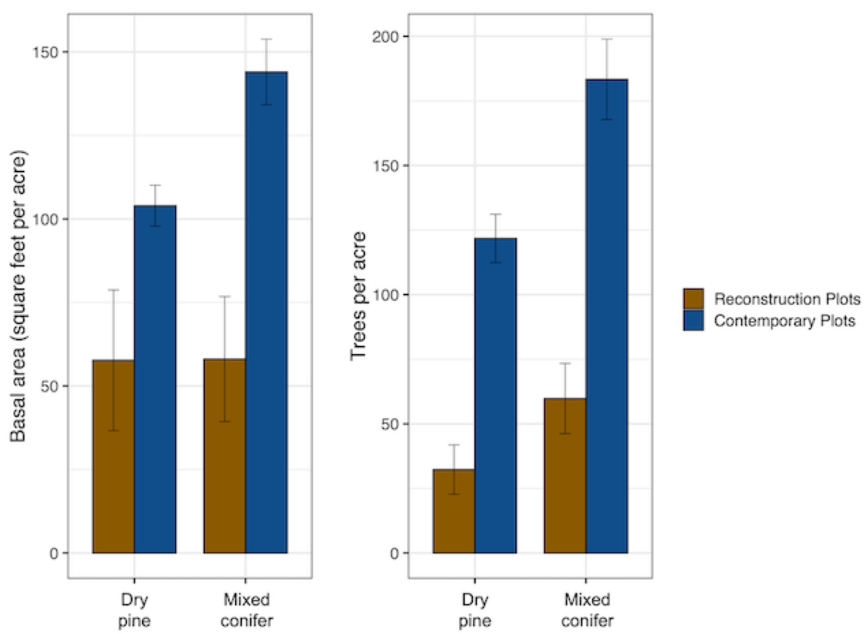



***Figure 4.1.*** *Reconstructions of historical (circa 1880) stands compared to contemporary (circa 2016) stands on the Malheur National Forest. Data is from Johnston (2017) and Johnston et al. (2018). Data is divided between dry pine and mixed conifer stands (see Section 2). Whiskers at the top of the bars indicated 95% confidence intervals for estimates.*

characterized by frequent, low severity fires and are today significantly more vulnerable to stand-replacing fire, drought, and insect disturbance.

Available evidence from the 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.

**Studies of change over time in eastern Oregon:**

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.

**Studies of change over time on the Malheur National Forest:**

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. (2021, 2018, 2017, 2016). This work demonstrates that xeric pine, dry pine, dry mixed conifer, and moist mixed conifer forest ecosystems on the MNF 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. As shown in Table 4.1 and Figure 4.1, there has been dramatic increases in the proportion of MNF stands composed of shade tolerant and less fire and drought tolerant species relative to shade intolerant and fire and drought tolerant species over the last 150 years.

| Species | 1860 basal area | 2015 basal area | % change |
|---|---|---|---|
| Western larch | 5.55 | 2.38 | -57% |
| Ponderosa pine | 41.46 | 63.97 | 54% |
| Douglas-fir | 1.45 | 14.87 | 925% |
| Grand fir | 2.35 | 57.50 | 2,346% |

***Table 4.1***. *Average reconstructed basal area (square feet per acre) of different species in unmanaged stands on the MNF in 1860 and in 2015. Source: Johnston 2017.*

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).

**Change over time in riparian areas and special habitats:**

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.

**Dissenting views:**

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 and Williams (2015), Williams and Baker (2012), and Baker (2012) use Government Land Office (GLO) records from the late $19^{th}$ 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."

## 4.2 Contemporary conditions

**Science synthesis about vulnerability of western US dry forests:**

*__Figure 4.2.__ Large patch of stand replacing fire following the Parish Cabin Fire that burned on the Malheur National Forest in 2012.*

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 associated with climate change driven drought (Parks and Abatzoglou 2020, Westerling 2016, Dennison et al. 2014). 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 much more severe than fires burning over the last two centuries (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).

**Contemporary conditions on the Malheur National Forest:**

Like other regions of the western US, the Malheur National Forest is today far more vulnerable to stand replacing disturbance. Approximately 18% of the Malheur National Forest has burned since 1985. Regional syntheses indicate that at least a third of the total area burned across the Malheur during this time has burned at high severity (Reilly et al. 2017). The true extent of high severity fire on the Malheur National Forest may be greater. Ongoing monitoring and research within several recent fires (the 2012 Parish Cabin and 2015 Canyon Creek Fires) suggests that more than 20% of trees that were alive in the immediate aftermath of fire subsequently died from fire damage or subsequent insect attack. Recent large fires on the Malheur National Forest have left very large (>1,000 acres) patches where all trees were killed (Figure 4.2). 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, almost all forests below 7,000 feet on the Malheur National Forest 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).

**Consequences of contemporary wildfires:**

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, recent research that evaluates fires in eastern Oregon (including fires on the Malheur National Forest) 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., in press). 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 through 8-25 years fire return intervals 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., in press).

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 on the Malheur National Forest has documented extensive invasion of grass species including cheatgrass and *Ventenata dubia* in stands burned at high severity. 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 are significant human costs to uncontrolled, high severity wildfire. 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 on the Malheur National Forest 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 on the Malheur National Forest may pose an even greater risk to communities in Grant and Harney Counties (Ager et al. 2021). The BMFP membership is unwilling to accept significant risk to life and property when restoration efforts can help protect lives and property, save taxpayer money in the long run, and help restore resilient forest ecosystems that are more capable of supporting native biodiversity and local communities.

**Consequences of drought and insects:**

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 (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 steep decline throughout the American West (Lindenmayer et al. 2012, Lutz et al. 2009, van Mantgem et al. 2009).

There is no existing comprehensive inventory of old tree mortality on the Malheur National Forest, but limited existing data suggests that the Malheur National Forest is experiencing roughly the same negative consequences from drought, disease, and insects as the rest of the West. Of particular concern are recent observations that suggest significant mortality of old-growth pine and larch across the Malheur National Forest as a consequence of 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 of the Malheur National Forest landscape will experience between 16-35% mortality of stand basal area in the next 15 years as a consequence of insect and disease (Figure 4.3). 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). **Just as the BMFP is unwilling to accept significant risk from wildfires when there are effective alternatives to reduce risk, we are unwilling to accept the loss of centuries old trees and the native biodiversity they support when restoration treatments that reduce competition can increase the survivability of these irreplaceable legacies** (Fettig et al. 2019).

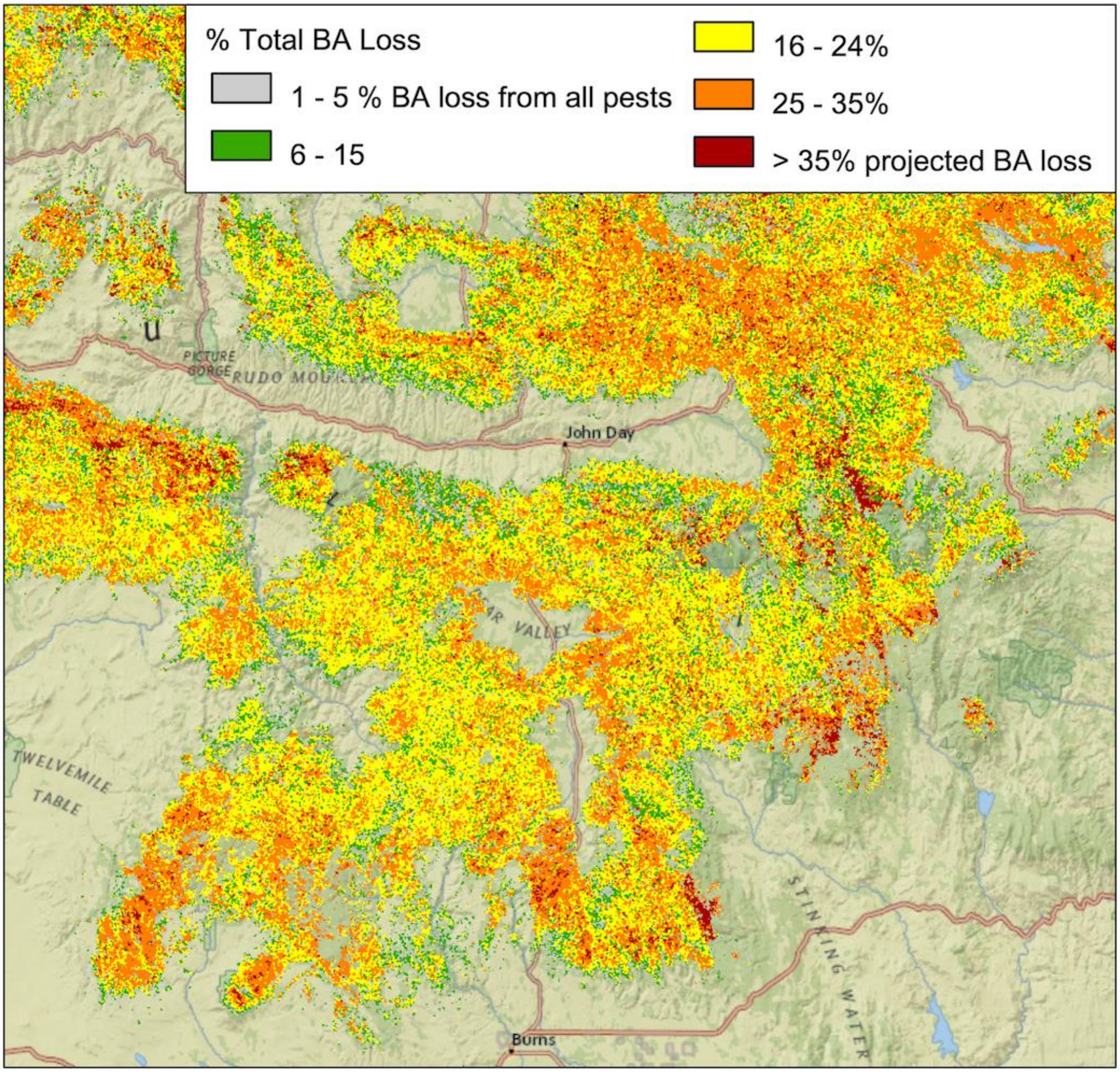



***Figure 4.3.*** *Projected basal area loss from insects and disease on the Malheur National Forest based on Forest Service insect and disease detection surveys. Source: National Insect and Disease Detection Survey (https://www.fs.fed.us/foresthealth/applied-sciences/mapping-reporting/detection-surveys.shtml).*

**Consequences to special habitats:**

A variety of special habitats are extremely vulnerable to current conditions. 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.

Quaking aspen (*Populus tremuloides*) is a one of the few hardwood tree species on the Malheur National Forest (MNF), and one of the only deciduous trees found outside of riparian areas. Aspen stands on the Malheur National Forest provide recreational opportunities and critical habitat for wildlife (Seager et al. 2015, Strong et al. 2010, Swanson et al. 2010, Seager 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 on the MNF (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 (Salabanks 2005, 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. 2013, Seager 2010, Swanson et al. 2010).

A rare and critically important habitat found on the Malheur National Forest 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 on the Malheur National Forest, although the dozen or so stands that exist on the forest are at the extreme edge of the native range of this species and many stands have experienced stand replacing fire. Many remaining stands are being encroached by fir.

## 4.3 Future conditions

**Climate change:**

Malheur National Forest 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 (Allen et al. 2009, Anderegg et al. 2013). 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).

**Climate change and special habitats:**

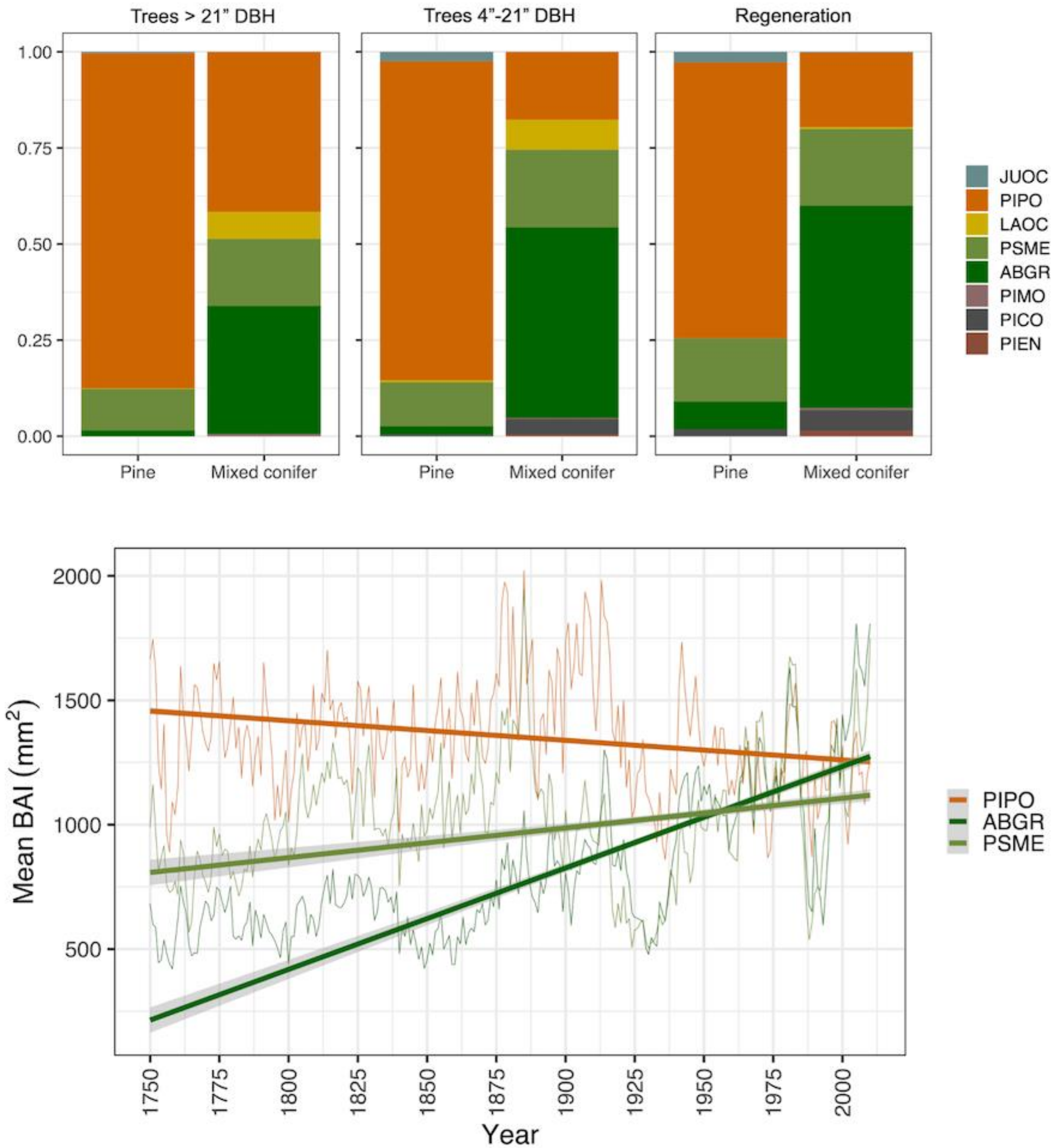
Directional climate change is expected to have profoundly negative consequences for special habitats on the Malheur National Forest. The increased frequency, duration, and severity of drought has resulted in widespread root mortality and crown loss in mature aspen stands in the Rocky Mountain region (Worrall et al. 2013). Drought associated with climate change is expected to result in significant new mortality of aspen across its current range, including much of eastern Oregon (Rehfeldt et al. 2009). 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).

**Successional trajectories**

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.

In summary, because they use more water, grow faster, and regenerate better in the absence of fire, shade tolerant grand fir is slowly replacing Malheur National Forest stands (Figure 4.4). However, grand fir cannot replace the ecological functioning of pine and larch. Pine and larch live much longer because their root architecture (a 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, Herman and Peterson 1969). 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).

The following page presents graphs demonstrating the trajectory of species composition in the Malheur National Forest.




***<u>Figure 4.4.</u>*** *Successional dynamics on the Malheur National Forest. Top panel: Regeneration by species and size class (regeneration = all trees <4" DBH). Data is divided between dry pine and mixed conifer (see Section 2). Note that a greater percentage of trees establishing over time is grand fir. Data from Forest Vegetation and Fuels (FVF) program. Bottom panel: Basal area increment of major conifer species over time. Note that ponderosa pine is declining relative to Douglas-fir and especially grand fir. Data from Johnston et al. (2016). Species codes: JUOC=western juniper, PIPO=ponderosa pine, LAOC=western larch (tamarack), PSME=Douglas-fir, ABGR=grand fir, PIMO=western white pine, PICO=lodgepole pine, PIEN=Engelmann spruce.*

# 5. Effects of restoration

It is critical that we evaluate the effects of restoration treatments in order to adapt treatments to better meet our goals and objectives. To judge the effectiveness of restoration treatments we are forced to rely heavily on retrospective studies from other areas because the vast majority of treatments on the Malheur National Forest were completed less than five years ago and many ecological responses to thinning and other treatments can a decade or more to characterize accurately (Watts et al. 2020, Lira et al. 2019).

However, both peer-reviewed literature about similar treatments in other areas and the available evidence from ongoing research and monitoring suggests that restoration treatments on the Malheur National Forest are achieving a number of the BMFP's goals. Of particular interest to the BMFP is evidence that: 1) treatments are moderating fire behavior and mitigating the risks 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) Malheur National Forest landscapes are more resilient and better adapted to future climate and disturbance stressors; and, 4) restoration actions are restoring special habitats.

## 5.1. Restoration treatments influence on fire behavior

**Science syntheses from across the western US:**

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. One study that summarized results of 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). Another comprehensive summary of Fire and Fire Surrogate results suggested that all treatments were relatively effective at moderating modeled fire behavior (Stephens et al. 2009).

A number of metanalyses and syntheses of fuel reduction projects across the American West, including Willms et al. (2017), Kalies and Kent (2016), Martinson and Omi (2013), Fulé et al. (2012), and Stephens et al. (2009) show that mechanical thinning followed by prescribed fire is generally effective at moderating wildfire severity. A few studies (e.g., Cram et al. 2015) report little difference in fire effects across a variety of treatments including thinning only and thinning followed by prescribed fire. But 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).

**Science results from the Malheur National Forest:**

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.

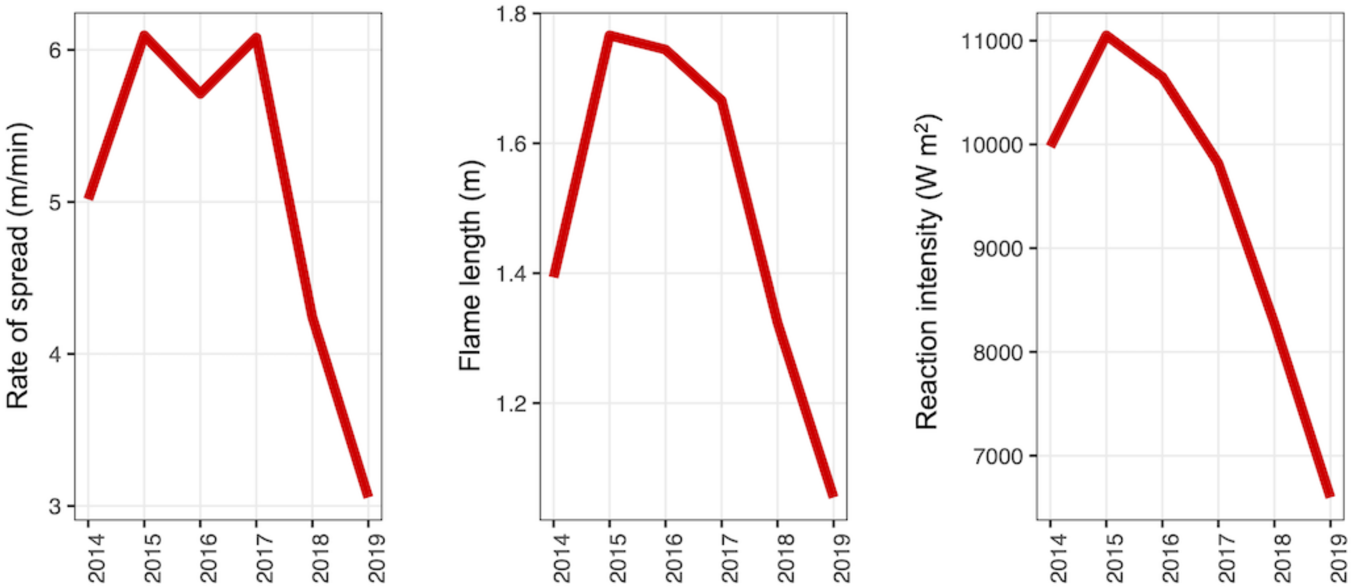



*__Figure 5.1.__ Modeled fire behavior parameters in treated units of the Marshall Devine restoration project in 2014 (before thinning) and in five years following thinning. Note that rate of spread, flame length, and reaction intensity increase for several years following treatment and then decline significantly. Data from Johnston et al. 2021.*

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).

## 5.2. Restoration treatment effects on diversity

**Science syntheses from the western US:**

Evidence 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).

**Science results from the Malheur National Forest:**

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 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. We identified 27 different genera of pollinators in thinned stands 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 on the Malheur National Forest 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).

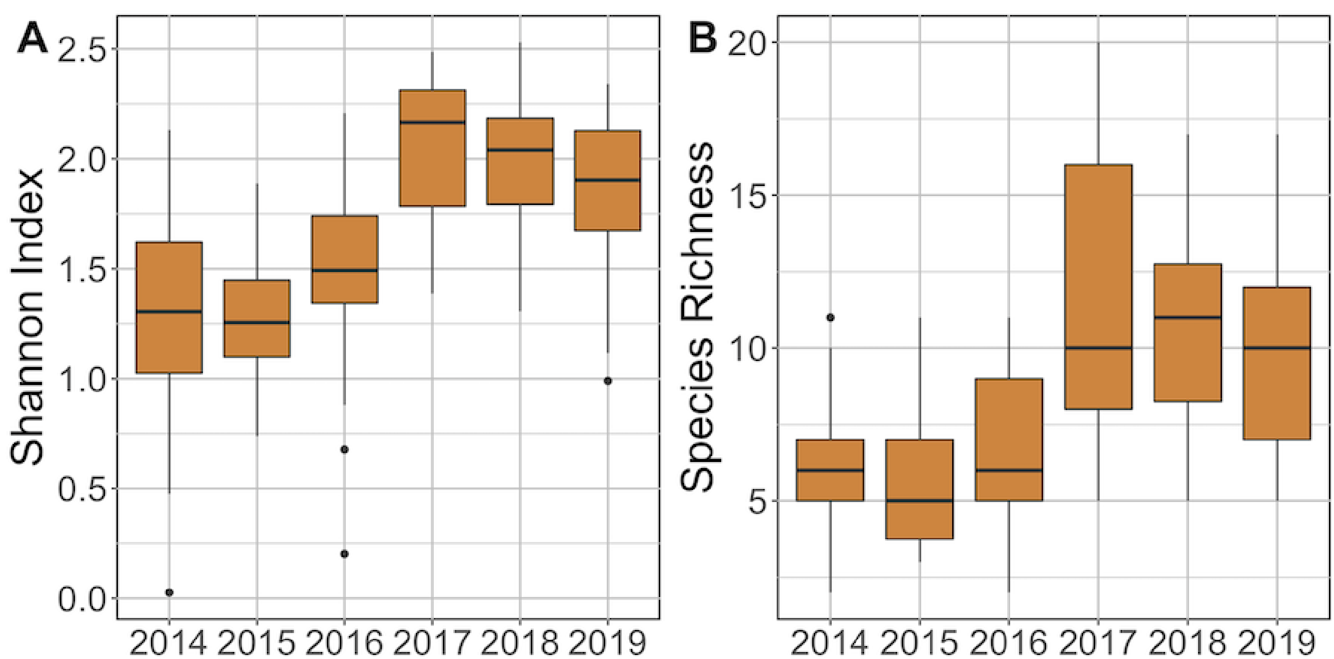



***Figure 5.2.*** *Two measures of vegetative diversity, Shannon's index (A) and Species Richness (B) before thinning of Marshall Devine treatment units in 2014 and for six years following thinning. Note that diversity declined slightly or remained the same immediately following thinning, but that diversity increased three years after thinning.*

Little or no peer-reviewed empirical research has been conducted on the Malheur National Forest that describes the effects of contemporary restoration treatments on the abundance and diversity of different wildlife species. 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 on the Malheur National Forest depends on inherent site variability (McIver et al. 2012).

## 5.3. Restoration treatments influence on forest resilience

A major goal of the BMFP is ensuring that treatments restore 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 to the BMFP 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). As noted in Section 3, 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 (Vernon 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 Forest Vegetation and Fuels team has collected data within the Marshall Devine planning area to determine if overstory trees are more vigorous following thinning that frees them of competition (Vernon et al. 2023). 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) (Figure 5.3).

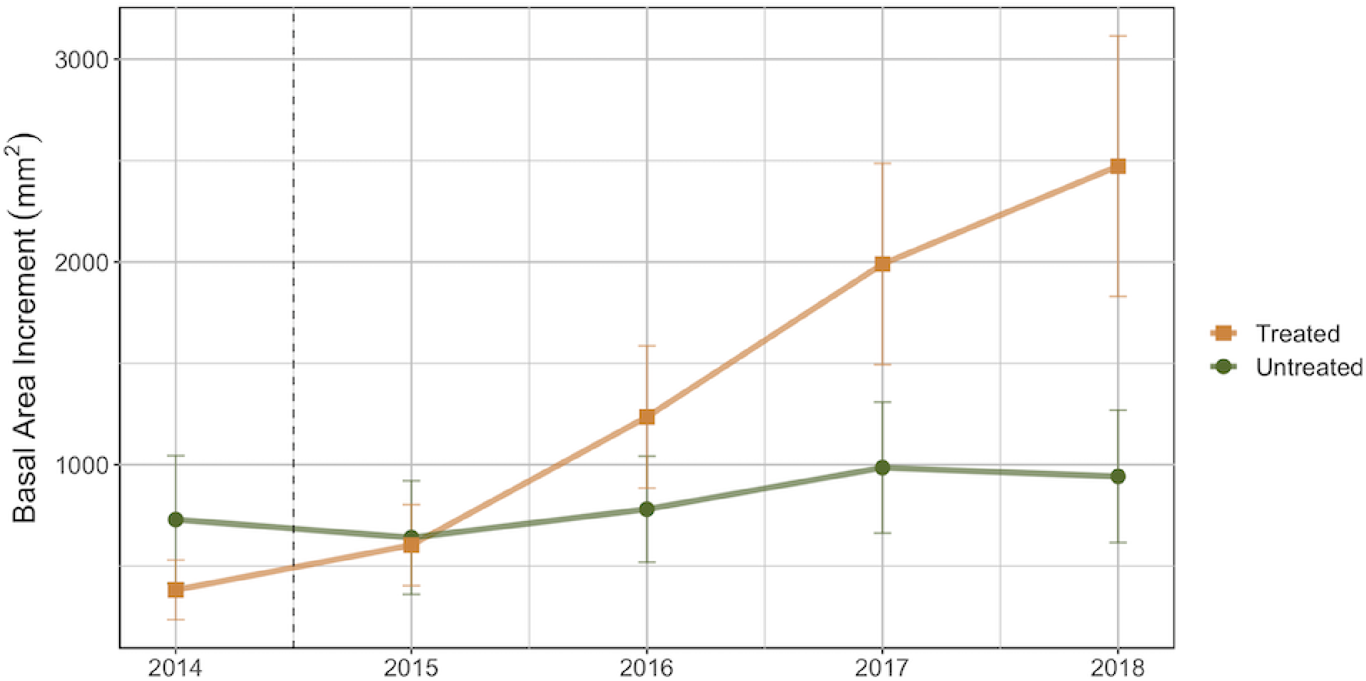



***Figure 5.3.*** *Average radial growth of trees over time in thinned and unthinned stands in the Marshall Devine planning area. Thinning occurred between the first and second year of measured growth (dotted line). Data from Vernon et al. 2023.*

### 5.4. Restoration actions and special habitats

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 (Jones et al. 2005, Seager 2010, Swanson et al. 2010, Seager et al. 2013, Seager 2017). 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).

## 6. Process-based restoration treatments

The goal of forest restoration is not to engineer a particular point-in-time forest condition, but to facilitate a range of desirable future forest responses to climate and disturbance processes. Drought, fire, insect attack and other perturbations are inevitable. Restoration treatments should be designed to adapt stands so that stands and landscapes will interact with these processes in such a way as to maintain key forest structures and continue to provide desired wildlife habitat, water quality, recreation, and other human uses.

The majority of the Malheur National Forest consists of upland forests. Below, we describe principles to guide upland forest restoration, the basic elements of upland forest restoration, and operational considerations for upland forest restoration in different forest types. Except in exceptional circumstances, the principles we describe should be applied whenever restoration is planned. Except in truly exceptional circumstances, it is unlikely that the BMFP would support projects that do not reflect these principles. The basic elements of upland forest restoration and the operational considerations for different forest types are designed to describe how these principles can be operationalized during upland forest restoration projects and are intended to provide managers with considerable flexibility in achieving the BMFP's goals and objectives.

Upland forest restoration will often be the most widespread type of treatment within different planning areas. It is the BMFP's intention that restoration treatments be strategic, take a landscape view, and integrate a variety of different types of restoration activities. To that end, we conclude by describing restoration of special habitats and riparian area restoration. We anticipate these treatments will most often occur in conjunction with upland forest restoration, although we support implementation of these activities independent of upland forest restoration. Management of wildlife, carbon stocks, and fire are also important considerations for upland forest restoration that we describe in Sections 7-9. We anticipate that wildlife management, management of carbon stocks, and fire management will be as tightly integrated with restoration treatments as possible, although we also acknowledge that planning for wildlife, carbon, and fire often requires a broader spatial and temporal perspective than many individual restoration projects.

## 6.1. Principles of upland forest restoration

We adapt basic principles of forest restoration presented in a variety of scientific syntheses that describe forest restoration activities in seasonally dry forests, including Hessburg et al. (2016), Stine et al. (2014), Agee and Skinner (2005), Brown et al. (2004), and especially Franklin and Johnson (2012) and Franklin et al. (2013). Our principles:

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

## 6.2. Elements of upland forest restoration

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

- Variable density thinning
- Openings
- 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 BMFP does not generally support removal of all trees across a large area unless these openings serve specific ecological restoration objectives.

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.

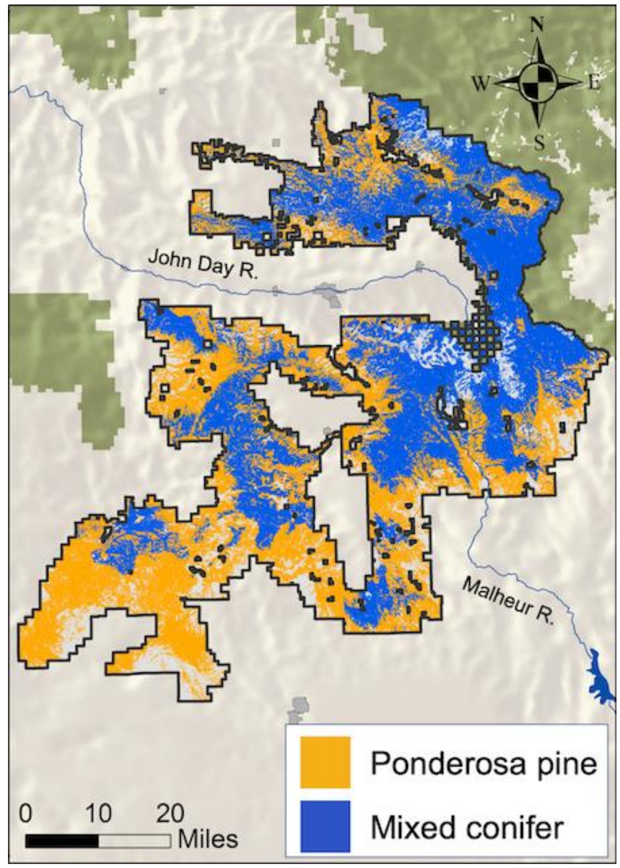



***Figure 6.1.*** *Approximate distribution of dry ponderosa pine and mixed conifer forests on the Malheur National Forest.*

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. As noted in Section 4, stands on the MNF are generally much denser than can be sustained over time. There has been significant conifer infill into meadows that were previously treeless, and into hardwood and riparian areas. 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 on the MNF.

## 6.3. Operational considerations for different forest types

**Vary treatments based on forest type and site potential:**

*Figure 6.2.* Twinflower and huckleberry indicate a moist mixed conifer site.

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 on the Malheur National Forest reflects differences in available soil water and atmospheric limits on transpiration (Figure 6.1; 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).

**Distinguish between stands with and without older grand fir:**

*Figure 6.3.* Old and decadent grand fir. Trees like this have significant value to wildlife and should always be retained in the course of silvicultural treatments.

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).

**Ponderosa pine–reduce basal area:**

In ponderosa pine stands, average stand basal area should be reduced to between 35 to 60 square feet of basal area. The BMFP strongly suggests that basal area targets be met at the scale of large treatment units, not on a per acre basis, meaning that if the basal area target is 50 square feet, we would expect that some acres would 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.

**Ponderosa pine–create fine-grained spatial pattern and openings where appropriate:**

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.

**Mixed conifer–reduce basal area:**

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, the BMFP strongly suggests that basal area targets in mixed conifer stands 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.

**Create fine, moderate, and coarse-grained spatial pattern:**

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). Figure 6.3 illustrates different silvicultural strategies and spatial pattern in different forest types.

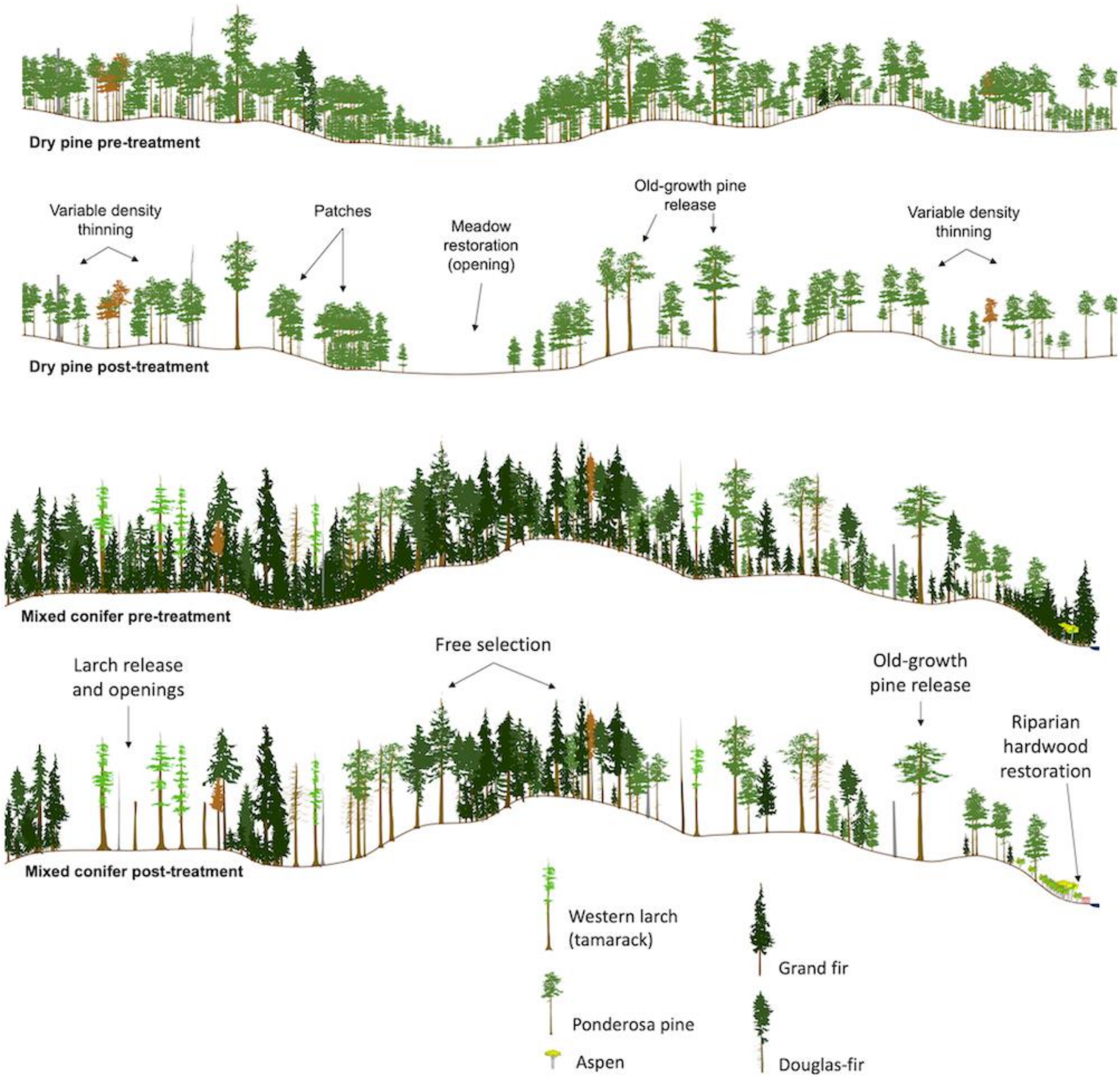



***Figure 6.3.*** *Conceptual rendering of pre- and post-treatment forest structure and composition following treatment in dry ponderosa pine (upper panel) and mixed conifer (lower panel) forests.*

**Utilize age-based tree conservation strategies:**

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).

**Select trees for retention with high wildlife value:**

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 (Figure 6.3).

**Treat as large of an area as practical:**

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.

**Create cost-effective restoration treatments:**

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.

**Use innovative and efficient contracting and implementation authorities:**

All restoration prescriptions should be flexible and tailored to the needs of particular sites. Using stewardship authorities, integrated resource contracts, designation by prescription, and other innovative contracting and implementation mechanisms can help achieve these goals.

| | Dry ponderosa pine | Mixed conifer |
|---|---|---|
| Overall spatial pattern | Generally fine grained except for created openings | Fine to coarse grained |
| Created openings | Where appropriate to restore meadow and savannah habitat. | Where appropriate to restore meadow and savannah habitat and to promote early seral shade intolerant species like larch and western white pine. |
| Leave patches | Small clumps of leave trees | Small to large clumps of leave trees |
| Mean diameter of residual trees | Significant increase from pre-treatment | Moderate increase from pre-treatment |
| Shade tolerant trees | Remove all or most except individuals established prior to the 1860s. | Retain all individuals established prior to the 1860s, as well as some younger individuals to replace older trees over time. |
| Residual basal area | 35-60 square feet per acre | 40-75 square feet per acre |

***Table 6.1.*** *Summary of differences in treatments between dry pine and mixed conifer stands*

## 6.4. Restoration of aspen

As noted in Section 4.2, 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. 2013, 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.

The BMFP strongly encourages treatments to maintain and expand aspen as part of the design of upland forest restoration treatments within individual planning areas. We also support treatments in aspen stands independent of upland forest restoration treatments when aspen stands are at high risk of being lost and it is not practical to integrate aspen restoration with upland forest restoration work. 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 on the Malheur National Forest (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 general, the BMFP encourages the removal of younger conifers while retaining older conifers. 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. 2013, 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 2010, Swanson et al. 2010, Seager et al. 2013, Keyser et al. 2005).

The persistence of aspen and the response of aspen to treatments can vary dramatically between aspen stands and the BMFP encourages 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. 2013, 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. 2013, White et al. 1998). The BMFP encourages fencing and other methods of excluding ungulates from aspen stands (such as jackstrawing felled trees or leaving coarse woody debris) as 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 2010, Seager 2013a). Beaver may browse aspen and fell overstory trees into perennial stream systems. Flooding that results from beaver dam construction can also enhance aspen habitat.

The BMFP strongly encourages the Forest Service to develop a comprehensive inventory of existing aspen stands and monitor the extent and decadence of aspen stands. Cloning and root sprouting can limit the genetic diversity, and where aspen are found in new areas following fire or other disturbance these stands should be fenced and protected from browsing (Lindroth and St. Clair 2013, Worrall et al. 2013, Swanson et al. 2010).

## 6.5. Restoration of white bark pine

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 on the Malheur National Forest 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 that the BMFP recommends for remaining stands on the Malheur National Forest 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.

## 6.6. Restoration of riparian areas and road and grazing management

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, Knopf et al. 1988). 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 to the BMFP 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). Other riparian restoration work that the BMFP supports includes actions to place large wood in streams or recruit future large wood to streams, placement of instream beaver dam analogues and other structures to create pools and other aquatic habitat components. The BMFP recognizes that some commercial products may result from operations in riparian areas, but the over-riding goal of work near streams, rivers, wetlands, and lakes should be ecological restoration.

Riparian restoration should be part of a whole watershed approach to restoration that integrates upland forest silviculture, recreation management, fire management, road management, and range management. Other actions that the BMFP supports in the context of whole watershed management include maintaining and stabilizing roads, relocating roads or closing roads administratively where appropriate where ensuring public access, work to improve stream crossings (including culvert replacement, repair and upgrading stream crossings, for instance, replacing culverts with bridges), conifer removal to stimulate forage, water developments to draw ungulates from riparian areas, and fencing of riparian areas and special habitats. A whole watershed approach will require creating strong partnerships with grazing permittees, recreationists, and other forest users.

### 6.7. Conservation of mature and old trees

Under development.

## 7. Fire Management

As noted in Section 4, forest of the Malheur National Forest are well adapted to low intensity surface fire, and many of the ecosystem services provided by the Malheur National Forests depend on fire. Indigenous communities used fires for thousands of years to manage natural resources (Armstrong et al. 2021, Roos et al. 2021). Prior to the advent of fire exclusion policies in the late 1800s, a combination of human and natural ignitions created a rich mosaic of resilient forest communities that were shaped by and sustained by fire. Fire created new habitat structures, stimulated new growth, helped cycle nutrients, and removed excess fuels (Turner and Gardner 2015).

Fire exclusion policies have led to a large contemporary fire deficit relative to historical conditions in western North American forests (Parks et al. 2015, Marlon et al. 2012). In the absence of low intensity surface fire that maintains open stands and removes fuel, communities, water supplies, and key ecosystem structures like old-growth trees are vulnerable to fast-moving high severity fire (Jones et al. 2018, Sankey et al. 2017, Williams 2013, Abella et al. 2007, Hessburg and Agee 2003). The extent and impacts from these fires will grow as the climate continues to warm (Parks and Abatzoglou 2020).

There are significant benefits to mechanical thinning that removes younger trees and provide resources necessary for old trees to thrive, even without reintroducing fire. Removing competition thinning protects these older trees from the effects of uncharacteristic drought, disease, and insect attack. Limited evidence from studies on the Malheur National Forest indicates that mechanical thinning in the absence of prescribed fire results in modeled fire behavior that is significantly less severe than untreated stands for up to a decade following thinning (Johnston et al. 2021). However, there is a consensus among scientists that only long-term solution to moderating the impacts of large high severity fire is using prescribed fire to reduce fuel loads during favorable weather conditions (Prichard et al. 2021, North et al. 2012). Despite the recognized importance of reintroducing fire to western landscapes, the use of prescribed fire is flat or declining across most of the western United States (Melvin 2020, Kolden 2019).

Smoke from wildfires has significant negative health effects. (Burket et al. 2021, Reid et al. 2016). But the huge quantities of smoke produced from wildfires has significantly more health effects than smaller volumes of smoke with shorter residence times that result from low intensity prescribe fire same dosage from other sources (Aguilera et al. 2021).

Given the importance of reintroducing fire to the Malheur National Forest to protect communities, the BMFP strongly supports a large increase in low intensity surface fire in order to prevent large and severe fire. This can be accomplished both by prescribing fire and making use of natural fire ignitions under moderate weather conditions to accomplish management objectives.

**Specific suggestions that will achieve these goals include:**

- Ensure funding is available to implement prescribed fire within mechanical treatment units.
- Develop partnerships with external workforces to complete prescribed fire, including but not limited to embracing a tribal co-management approach to fire use, increasing training opportunities, and developing effective agreements with private contractors to apply fire to the Malheur National Forest landscape.
- Consider developing a landscape scale programmatic NEPA analysis that covers prescribed fire and fire use.
- Develop a publicly available database of areas treated with fire and plans for future fire use.
- Develop partnerships with regulatory agencies to lengthen burn windows.
- Use wildland fire when appropriate to meet management objectives. The effective use of wildland fire can be facilitated by forward looking planning that identifies potential operational delineations (PODs), or lines suitable for controlling wildfire and allowing wildfire to accomplish management objectives where appropriate within those delineations (Figure 7.1). Using wildland fire to achieve resource objectives can best be accomplished by close coordination between Malheur National Forest silviculturists and fire managers.

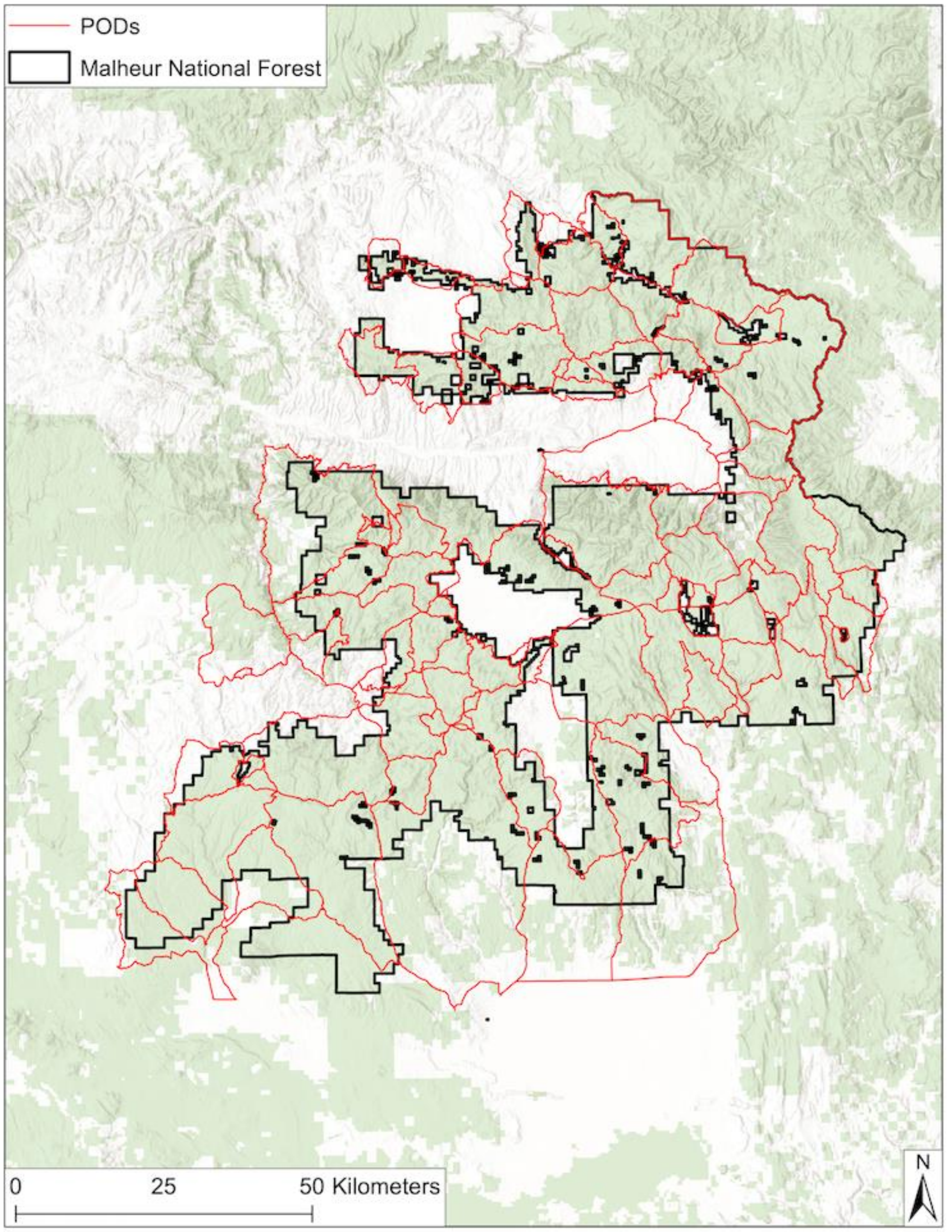
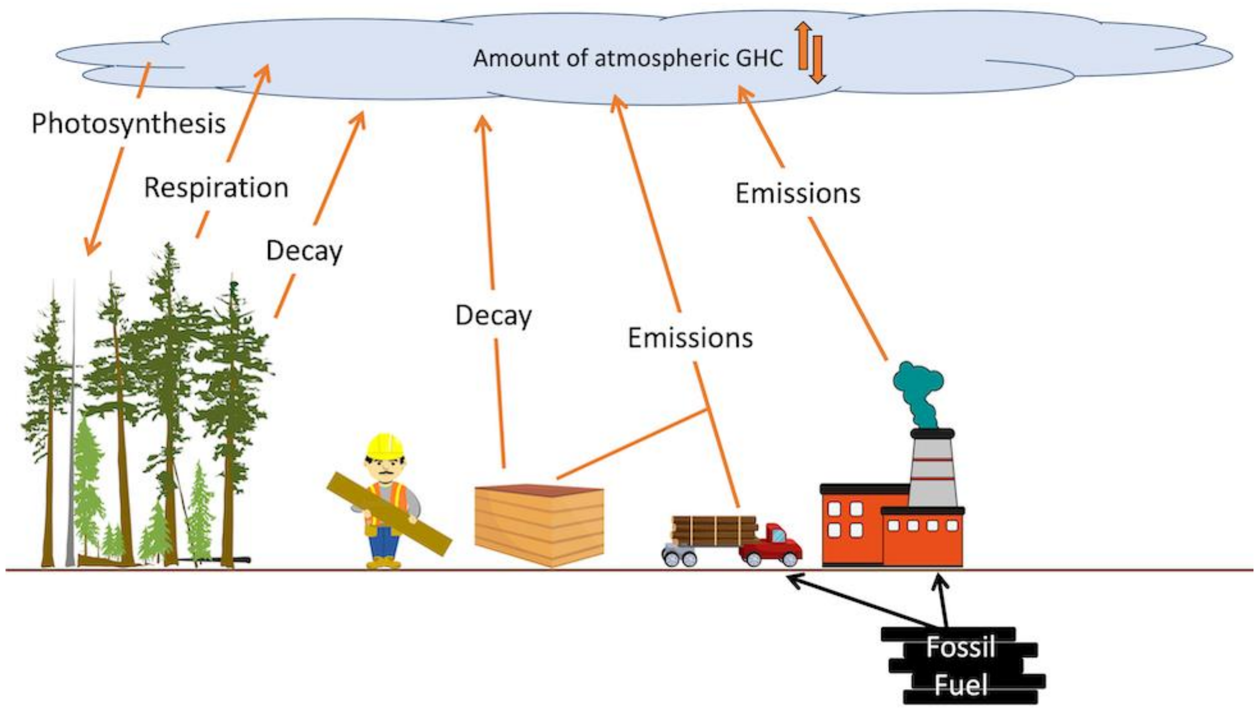



***Figure 7.1.*** *Potential Operations Delineations (PODs).*
*Red lines indicate areas highly suitable for fire containment.*

# 8. 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 products are 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 Malheur National Forest stores marginally more carbon on an annual basis than is lost through decomposition and disturbance. Annual carbon storage on the Malheur is significantly lower than typical national forests in the Pacific Northwest, because forests on the Malheur 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 on the Malheur 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 extent of high severity fire. There has been no empirical research that quantifies the effects of different active management strategies on carbon stocks on the Malheur National Forest, 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 on the Malheur NF 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. 2018, 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 (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 (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 on the Malheur NF 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 Malheur NF 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 Malheur NF 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 on the Malheur NF, 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 2021, Lehmkuhl et al. 2007).

## 9. Monitoring and adaptive management

[UNDER DEVELOPMENT]

## 10. Conclusion

The Blue Mountains Forest Partners (BMFP) is a diverse group of stakeholders. Each of these agreements has been hard won based on all party's willingness to look at the science, and each other's needs and values. While these are not binding on the Forest Service, we hope that they are useful. We believe that they are a good balance of ecological, economic, and social needs. We also believe that they represent an up to date synthesis of the relevant science that is more than sufficiently detailed to be used in drafting NEPA documents that will survive judicial review.

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