CLIMATE CHANGE AND WILDFIRE: IMPLICATIONS FOR FOREST MANAGEMENT IN THE BLUE MOUNTAINS OF EASTERN OREGON

by

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Climate change and wildfire: implications for forest management in the Blue Mountains of eastern Oregon

Thesis directed by Associate Professor Joel Hartter

Shifting climate and wildfire regimes are changing forest structure and function globally. In the western US, future forest structure will be determined by interactions between climate change and disturbance, including increasingly frequent large wildfires, as well as the forest management actions of landowners and managers. While research on the ecological impacts of these changes is rapidly expanding, there is limited focus on how forest vulnerability may vary across land ownerships, which may have varying capacities for climate-adaptive forest management. In this dissertation, I explored how coniferous forests in the Blue Mountains of eastern Oregon are vulnerable to climate and wildfire interactions across land ownerships, and investigated the adaptive capacity of private forest owners. First, I used LANDIS-II, a dynamic forest landscape model, to simulate potential impacts of climate change and wildfire on tree species establishment, abundance, and growth. I found that, despite establishment declines in moisture-limited areas, drought- and fire-tolerant ponderosa pine and Douglas-fir expanded their distributions under high wildfire activity and climate change scenarios, while less tolerant species such as subalpine fir declined. Second, I surveyed 184 sites across eight burned areas 15-21 years post-fire to understand how topography, climate, and post-fire legacies influence juvenile conifer abundance. One-third of sites contained no juvenile conifers, potentially indicating regeneration failure on warm slopes at low elevations far from a post-fire seed source. However, juvenile conifer abundance in most high elevation sites exceeded recommended stocking levels, suggesting forest resilience in high elevation forest types. Finally, I interviewed 50 private landowners to gauge their capacity for climate change adaptation. Very few landowners adapted to climate change intentionally, in part due to climate change skepticism.

However, many forest owners implemented incidental adaptation actions, including fuels reductions, motivated by factors such as wildfire risk mitigation. Ultimately, private forest owners require educational, financial, and operational support to engage in climate-adaptive forest management. Just as the impacts of climate change and wildfire vary by forest type in the Blue Mountains, adaptation recommendations must reflect the varying adaptive capacities of local landowners and managers.

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CHAPTER I

INTRODUCTION

1.1 Climate-wildfire interactions alter forest structure

Climate change impacts forest ecosystems globally. Rising temperatures and shifting moisture regimes alter tree species distributions through gradual shifts in growing conditions (Coops and Waring 2011, Zhu et al. 2012, Campbell and Shinneman 2017). More immediately, climate-mediated changes in the frequency and size of wildfires may rapidly change temperate forest structure (Turner 2005, Enright et al. 2015, Vose et al. 2015). Rising global temperatures are lengthening fire weather seasons (Jolly et al. 2015), increasing the probability of large, high severity wildfires in many forested regions including western North America where fire has been suppressed since the late 19th century. As a result, the frequency and size of large wildfires in the western United States is projected to increase (Barbero et al. 2015, Liu and Wimberly 2016, Westerling 2016). Human ignitions are also shifting fire regimes and lengthening fire seasons (Balch et al. 2017). Humans ignited four times as many large wildfires (defined as the largest 10% of fires for a given ecoregion) as lightning in the United States from 1992-2015 (Nagy et al. 2018). These trends raise questions about whether forested landscapes are resilient to shifting wildfire and climate regimes, as well as how climate-adaptive forest management may conserve forest ecosystem function and resulting ecosystem services for communities at the wildland urban interface.

Resilient forest ecosystems can maintain a similar structure and function following disturbance (Holling 1973). In complex adaptive systems theory, ecosystem state transitions occur when ecosystems lose resilience and are pushed into an alternate state by a sufficiently strong shock (Holling 1973). Accumulating evidence suggests that high severity wildfires followed by warmer, drier conditions may lead to ecosystem state transitions, in which forests are replaced by alternate vegetation communities that persist for decades or centuries (Scheffer et al. 2001, Lenihan et al. 2008, Biggs et al. 2009). Several studies suggest that climate change and wildfire are interacting in this way to produce ecosystem shifts in the western US, where high severity wildfire and drought have shifted historically-forested areas to shrub or grassland cover (Williams et al. 2010, Savage et al. 2013, Rother and Veblen 2016, Stevens-Rumann et al. 2018). Examples include observations from the Klamath Region of northern California and southwest Oregon, where shrubs and broadleaf trees quickly regrow following high severity fire and outcompete establishing conifers on many sites (Tepley et al. 2017). The shrub ecosystem can be maintained when fires repeatedly reburn the same area, which may increase in likelihood under warming conditions, killing newly established conifers. In Colorado, large patches of high-severity fire in ponderosa pine forests persist as grassland up to 15 years post-fire on hot, dry sites where seed availability is low (Rother and Veblen 2016). Research on the biophysical drivers of regime shifts in forested landscapes is growing (e.g. Puhlick et al. 2012, Dodson and Root 2013), as well as studies explicitly considering how landowners and managers can adaptively manage forests to modify or ameliorate projected changes in forest structure (Keenan 2015, Halofsky et al. 2018a, 2018b).

Many recommendations made by scientists are directed towards agency personnel managing forest structure on public forestlands. However, models of projected impacts of climate change on forests in the western US suggest the greatest vulnerability to regime shifts may exist at low elevations where heat and drought are most extreme (Vose et al. 2015). In the western US, much of the federally-managed forest land is located at high elevations, some of which is designated as wilderness. In contrast, low elevation conifer forests are characterized by mixed ownership. In addition to state and local forestlands, large portions are privately owned, and therefore adaptive management recommendations must reflect the capabilities and limitations of private forest owners. In my dissertation, I investigate the interface between the changing forest ecology of the Blue Mountains ecoregion in the inland northwest US, and the perceptions and responses of private forest owners.

1.2 Changing dynamics in dry and moist mixed-conifer forests

Large wildfire activity in the western US increased markedly in the mid-1980s primarily due to two factors: forest management and fire suppression policies, and climate change-induced increases in spring and summer temperatures (Marlon et al. 2012, Dennison et al. 2014, Hamilton et al. 2016, Westerling 2016, Schoennagel et al. 2017, Crockett et al. 2018). In the inland northwest, a region with one of the largest deficits of high severity fire compared to historic fire regimes (Tracy et al. 2018), fire suppression, logging, and grazing have altered forest structure and led to stand densification in formerly patchy forests (Hessburg et al. 2015). Selective logging halved the density of old-growth, fire-resistant trees, and shade-tolerant trees have replaced them in moist areas (Merschel et al. 2014). Thirty to forty percent of forests in the Blue Mountains are denser than their natural range of variability (Haugo et al. 2015, DeMeo et al. 2018).

According to tree-ring studies, from the 17th century to 1900, historic fire regimes in mixed-conifer forests of the inland northwest were characterized by frequent, low-severity fires occurring every 15-40 years (Heyerdahl et al. 2001). Dry mixed-conifer forests at low elevations had a frequent low-severity fire regime similar to ponderosa pine forest, but that frequency declined and severity increased with elevation (Merschel et al. 2014). In four watersheds in the Blue Mountains ecoregion, sixty percent of fires pre-1900 were greater than 250 ha in size, while there were no fires greater than 250 ha in size documented through the 20th century (Heyerdahl et al. 2001). This transition is attributable to the arrival of Euro-American settlers in the late 19th and early 20th centuries, in addition to the implementation of fire suppression beginning effectively in the 1940s. Overgrazing by livestock and ungulates reduced flashy fuels, which historically facilitated fast-burning low-severity ground fires, contributing to infill of formerly patchy dry conifer forests with dense shade-tolerant species (Hessburg et al. 2005). Fire exclusion has had a similar effect, creating a more homogenous forest with greater vertical and horizontal fuel continuity (Agee 1998, Hessburg et al. 2005).

In the northwestern US large wildfires historically occurred when warm, dry winters (El Niño years) and persistent drought conditions occurred sequentially (Wright and Agee 2004). The El Niño-Southern Oscillation and Pacific North-American (PNA) pattern reinforces or counteracts the effects of anthropogenic warming at multidecadal time scales (Abatzoglou et al. 2014). Climate change is projected to increase temperatures by 3°C to 5.5°C by the end of the 21st century. Precipitation projections are more variable, but models generally predict 5-15% more precipitation in winter and 5-10% less precipitation in summer (Mauger and Mantua 2011, Rupp et al. 2017). Though winter precipitation may increase, more of it will fall as rain as increased average winter temperatures push the freezing line to higher elevations. This is predicted to result in a 70% decline in April 1st snowpack, with the date of 90% snowmelt occurring up to one month earlier (Mauger and Mantua 2011). Lower and earlier spring runoff results in decreased average summer moisture storage, and temperature increases will result in higher summer evapotranspiration and drier forests, increasing the probability of fire ignition and spread (Mauger and Mantua 2011). Rising temperatures are also projected to alter tree species distributions.

Tree species distribution models and process-based simulations project several climate change-induced changes in vegetation in the inland northwest. At high elevations, subalpine forests may entirely disappear in the Blue Mountains over the 21st century under rapid climate change, while at low elevations, large areas of ponderosa pine forest may convert to sagebrush shrublands (Kim et al. 2018). Dynamic global vegetation models (DGVMs) project a general upward movement of the forest-steppe ecotone in latitude and elevation (Kerns et al. 2018). Wildfire will potentially facilitate transitions from forest to non-forest or different forest types, impacting wildlife habitat and ecosystem services. Recommendations for climate- and fire-adaptive forest management include reducing tree densities through mechanical thinning and prescribed burning, expanding wildland fire use, and enhancing heterogeneity across forest landscapes (Hessburg et al. 2015, Keenan 2015, Halofsky et al. 2018b, Walker et al. 2018). Whether climate-wildfire interactions have positive or negative impacts on ecosystems and communities will depend partly on our understanding of shifting dynamics, as well as management decisions and actions.

1.3 Research aims

In my dissertation I investigate the effects of climate and fire on forest ecosystems, including post-fire tree regeneration, and the factors influencing management responses of private landowners. I aim to determine how the vulnerability of forest ecosystems and communities in the wildland urban interface, comprised of their exposure and sensitivity to climate and fire shifts, may be moderated by the adaptive capacity of forest owners and managers. My research bridges the ecological and social science literatures on averting ecological regime shifts (e.g. Biggs et al. 2009) and overcoming barriers to climate adaptation (Moser and Ekstrom 2010).

In Chapter 2 I present a forest landscape simulation model through which I investigate the effects of potential future climate change and wildfire scenarios on forest structure and composition within a subregion of the Blue Mountains Ecoregion. I identify how species establishment probabilities change under three future climate scenarios, and investigate the effects of climate change and wildfire interactions on species biomass and abundance across land ownerships. This study demonstrates a spatially explicit method for determining landowner-specific priorities for climate-adaptive forest management. In Chapter 3 I investigate the degree to which regional climate impacts post-fire tree regeneration when accounting for site-level variables. This study aims to determine whether fire-catalyzed ecosystem transitions are currently happening in the inland northwest. The site level variables I analyze include burn severity and distance to nearest potential seed source, as well as ground cover and topography. In Chapter 4 I evaluate the adaptive capacity of private non-industrial forest owners, specifically drawing connections between cognitive and resource-related barriers to climate-adaptive forest management. This research aims to inform policy, education, and outreach interventions that enhance the ability for private landowners to mitigate risks posed by the climate-wildfire dynamics explored in Chapters 2 and 3. Lastly, in Chapter 5, I summarize the implications of my work for forest owners and managers, as well as scientists developing tools and guidance for managers. I also discuss future research directions and priorities, including expanding the concept of resilience in forest management as ecological and biophysical baselines continue to shift under changing climate and wildfire regimes.

CHAPTER II

SIMULATED EFFECTS OF WILDFIRE AND CLIMATE CHANGE ON TREE SPECIES: MANAGEMENT IMPLICATIONS VARY BY FOREST OWNERSHIP

2.1 Abstract

Tree species distributions are expected to shift in response to climate change. These shifts may be gradual due to the long life spans and dispersal limitations of trees, or rapid if disturbances and climate interact to abruptly alter post-fire species composition and forest structure. Few studies explore how the effects of fire-climate interactions vary by land ownership[1], and in the inland northwest, US, public and private lands generally occupy different elevation ranges and dominant forest types. I used a forest landscape simulation model to perform a landowner-specific analysis of projected changes in forest structure and composition on private, State, Bureau of Land Management, and National Forest land in eastern Oregon. I implemented three future climate scenarios, including historical climate as well as warming scenarios based on Representative Concentration Pathways 4.5 and 8.5. I crossed these climate scenarios with two fire scenarios, including contemporary fire behavior and more frequent large wildfires. Species establishment probabilities, biomass and abundance declined for some upper montane and subalpine species under climate change, with the largest potential impacts on

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[1] Throughout this Chapter I use the terms "ownership" and "management" to refer to forestland tenure. I use "management" where appropriate, recognizing that federal public lands are managed by federal agencies but technically owned by the American public. I use "ownership" when referring wholly or in part to private forestlands.

private and National Forest lands. While ponderosa pine and Douglas-fir increased in biomass and abundance under all scenarios, primarily due to enhanced establishment and growth at high elevations, they were projected to experience declining establishment probabilities at low elevations on BLM and State-managed lands in the study area. The high wildfire scenario led to larger declines in vulnerable high-elevation species under climate change, and also produced greater variability in biomass trajectories through time. This study demonstrates how spatially explicit modeling experiments can aid forest managers in identifying local priorities for climate-smart[2] forest management, and can potentially inform an "all lands" approach to regional climate change adaptation. Additionally, private forest owners may manage forest types as diverse as those on public lands, and therefore may need to consider diverse strategies for climate-adaptive forest management.

[2] "The intentional and deliberate consideration of climate change in natural resource management, realized through adopting forward-looking goals and explicitly linking strategies to key climate impacts and vulnerabilities." (Stein et al. 2014)

2.2 Introduction

Aridity and fire activity driven by climate change in western North America are projected to alter forest composition and structure (Coops and Waring 2011, Enright et al. 2014, Dobrowski et al. 2015, Mathys et al. 2017). Indeed, high severity wildfires and drought conditions may already be changing post-disturbance recovery trajectories and tree growth in some areas (Rother et al. 2015, Harvey et al. 2016, Rother and Veblen 2017, Stevens-Rumann et al. 2018). Therefore, assessing where forests are most vulnerable to the impacts of climate change and wildfire is an emerging research area that is critical to informing management interventions (Chmura et al. 2011, Buma and Wessman 2013, Sample et al. 2014, Millar and Stephenson 2015, Halofsky et al. 2016b, 2018a, 2018b, Schoennagel et al. 2017).

Both public and private forest managers and landowners report that one of the central factors impeding climate change adaptation in forest management is uncertainty surrounding future local impacts (Grotta et al. 2013, Kemp et al. 2015, Boag et al. 2018). Currently, multiple tools exist to project future impacts on tree species, including bioclimatic envelope models, a widely used form of species distribution model. Bioclimatic envelope models use data on species locations and historical climate to identify climatic predictors of species presence, then project where species ranges may expand or contract under future climate change scenarios (Heikkinen et al. 2006). However, the degree to which tree species will shift to suitable climate regions depends on dispersal, competition, and disturbance patterns, all of which may also be altered by climate change (Littell et al. 2011, Campbell and Shinneman 2017). Forest landscape models (FLMs) aim to incorporate these factors. FLMs are spatially explicit and incorporate climate, disturbance, species interactions and life history traits through time, permitting mechanistic assessments of climate change impacts on forests. Therefore, FLMs present an opportunity to reduce uncertainty surrounding region-specific impacts of climate change and wildfire.

The spatially explicit nature of FLMs also enables species vulnerability assessments for specific land ownerships within a study area. I argue this is an important avenue for future research because of calls for an "all lands approach" to forest management. The US Department of Agriculture Forest Service advocates for an all lands approach acknowledging that both the public benefits of forests and threats to forests do not recognize property boundaries. The USDA Forest Service 2012 Planning Rule directs managers to consider the broader landscape context in planning, assessing and monitoring management actions (USDA Forest Service p. 21164). Forest Service officials also recognize the importance of an all lands approach to climate change adaptation:

> "The effects of climate change cross borders and boundaries; no single one of us can succeed alone. Last year, Secretary of Agriculture Tom Vilsack offered a broad vision for an all-lands approach to sustaining and restoring the nation's forests. That means using all USDA resources and authorities, in collaboration with NRCS, to sustain the entire matrix of federal, state, tribal, county, municipal, and private forests... Restoration requires a strategy for addressing climate change, and the Forest Service has formulated a Strategic Framework for Responding to Climate Change. Our strategy is predicated on working through alliances with state, private, tribal, and other partners on a landscape scale. How can we work together, based on sound science, to help ecosystems adapt to the effects of climate change?"
> –Tom Tidwell, Chief, US Forest Service (Tidwell 2010)

Rapid warming in the northern hemisphere will continue in the coming decades, with global temperature increasing by 1.5°C by mid-century unless coordinated global action is taken to dramatically reduce emissions (IPCC 2018). Unfortunately, no such actions are underway. Therefore, an all lands approach to forest management will likely become more important through the 21st century. For example, if certain tree species or forest types are vulnerable to climate change-induced extirpation within a region, different land managers could coordinate to maintain multiple refugia and habitat connectivity through time, representing a resistance approach to climate change adaptation. Alternatively, managers could coordinate across ownerships to enhance the resilience of forests to changing climate and fire regimes so that they recover their general structure and function following wildfire. Finally, in areas where forest will inevitably transition to another ecosystem type, managers might elect to facilitate those transitions. Where and when managers adopt any of these three adaptation approaches will depend on the values of stakeholders and specific management goals.

The first step towards an all lands approach to adaptation involves developing regional adaptation strategies (Raymond et al. 2014). At the same time, it is important to acknowledge that state, federal, and private forest managers have different adaptive capacities. Federal land managers have greater access to expertise in research, planning, and funds for implementation than many state managers and private landowners. Landowner-specific vulnerability information may help agencies and supporting organizations such as University extension services provide resources for emerging management issues. Below I demonstrate how FLMs can provide actionable region-specific information through a case study in eastern Oregon.

2.2.1 Case study: tree species vulnerability in eastern Oregon

Climate change is projected to change temperature and precipitation in eastern Oregon considerably (Halofsky and Peterson 2016). Given a range of projections, by the 2080s winter temperatures will be on average 3.3˚C warmer, and 5˚C warmer in the summer (Mauger and Mantua 2011). Precipitation projections are more variable, but models generally predict more precipitation in winter (15% higher) and less in the summer (17% lower) (Mauger and Mantua 2011). Though winter precipitation may increase, it may fall as rain more often than snow as rising average winter temperatures push the freezing line to higher elevations. The result may be a 70% decrease in April 1st snowpack, with the date of 90% snowmelt occurring up to one month earlier (Mauger and Mantua 2011). Less spring runoff occurring earlier reduces average summer moisture storage, and temperature increases will result in higher summer evapotranspiration (Mauger and Mantua 2011). The maximum number of consecutive dry days (precipitation <1 mm) a year is projected to increase by 5 to 10 days (Vose et al. 2012). Taken together, these projections suggest forests will be moisture stressed and susceptible to insect and disease outbreaks (Halofsky and Peterson 2016).

Large wildfire activity in eastern Oregon increased suddenly and markedly in the mid-1980s with higher frequencies of large wildfires, longer wildfire durations, and longer fire seasons (Hamilton et al. 2016). Across the West, large wildfire activity is strongly associated with increased spring and summer temperatures and earlier spring snowmelt (Westerling 2016, Crockett et al. 2018). In the northwest US, large wildfires historically occurred when warm, dry winters (El Niño years) and persistent drought conditions occurred sequentially (Wright and Agee 2004). Given the current trajectory of global greenhouse gas emissions, climatologists predict further temperature increases will raise the frequency and severity of drought in the western US, further increasing the frequency and size of wildfires (Barbero et al. 2015).

In this study, I use the LANDIS-II forest landscape model (Scheller et al. 2007, He 2008) to inform local climate-adaptive forest management. Specifically, I identify where conifer forests may be vulnerable to establishment limitation under changing climate, as well as how climate and wildfire interactions may impact species biomass and abundance across a multi-ownership landscape in eastern Oregon. I hypothesized that 1) given climate change scenarios, species establishment probabilities for drought-intolerant conifer species will generally decline; and 2) given that low elevations may experience stronger heat and drought effects, public and private land ownerships at low elevations will experience greater declines in species establishment probabilities, while high-elevation National Forest land may be more resistant to changes; and 3) scenarios with greater projected warming and high fire activity will result in the largest relative declines in species extent and biomass.

2.3 Methods

2.3.1 Study Area

The Blue Mountains are located in the inland northwest United States and have a history and culture of ranching, agriculture and logging. The local timber industry collapsed in the early 1990s, impacting the socioeconomic fabric of local communities; however, several industrial logging companies still operate locally and some non-industrial private forest owners have commercial timber operations (Christoffersen 2005). The study area contains the Heppner Ranger District and a portion of the North Fork John Day Ranger District of the Umatilla National Forest (Figure 2.1). I selected this region in part because it features a dramatic soil and climatic moisture gradient from southwest to northeast, resulting in a wide diversity of plant community types.

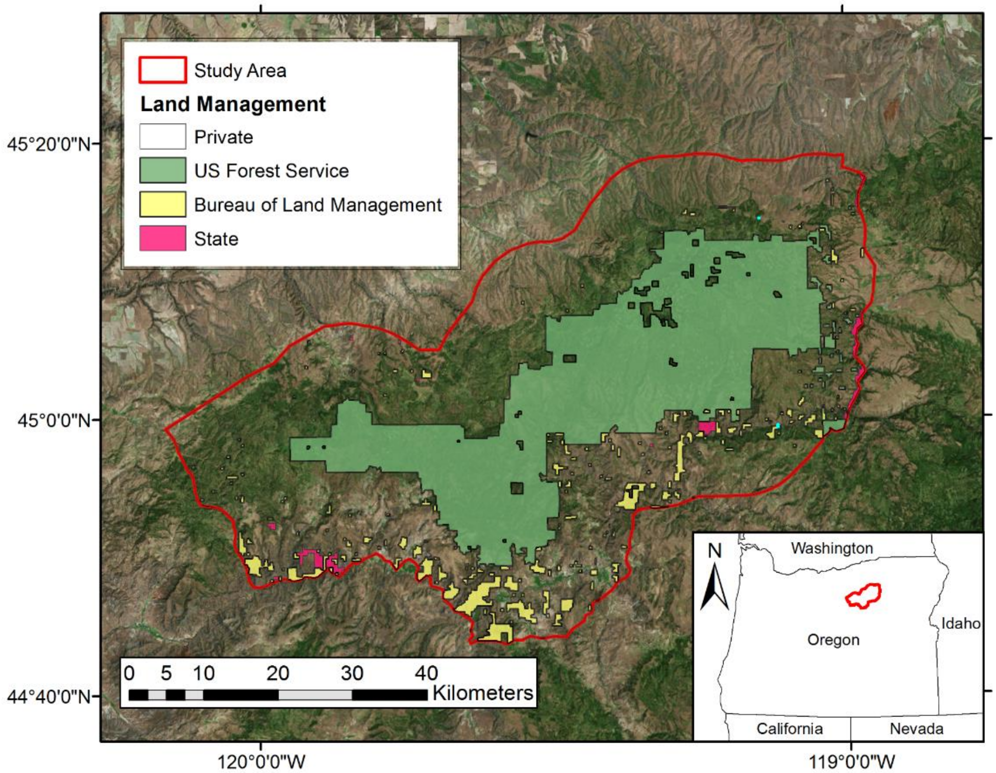

Figure 2.1. Study area in eastern Oregon showing land management classes. US Forest Service land is primarily comprised of the Heppner Ranger District of the Umatilla National Forest.

Within the study area, the US Forest Service (USFS) manages 55% of forestland, while the Bureau of Land Management (BLM) manages 6% and State of Oregon lands comprise <1%. Despite state lands comprising a small portion of the study area, I saw no reason to exclude them. State lands comprise only 3% of forestland statewide, but state foresters nonetheless experience climate change and wildfire impacts on their forests. Private corporate owners control 10% of forestland in the study area, and private nonindustrial landowners manage the remaining 28% (Table 2.1). Both types of private forest owners must comply with the Oregon Forest Practices Act (OFPA). While the OFPA has no climate change-specific management regulations, those relevant to climate-adaptive forest management include a requirement to replant within two years of harvest yielding a minimum density of 100 to 200 trees per acre in eastern Oregon. The

OFPA instructs forest owners to plant trees that are genetically adapted to the planting site (OFPA 2018).

Table 2.1. Proportion of the study area occupied by each land management class.

Land Management	Total Area (ha) (% of forestland)
Private	230,452 (55%)
US Forest Service	112,269 (38%)
Bureau of Land Management	13,374 (6%)
State	1,865 (0.5%)
Total Area	357,960

I bounded the study area by major highways on the east and west side of the National Forest because highways potentially reduce the probability of wildfire spread from adjacent forested areas. In the absence of major roads on the north and south sides, I bounded the area using a 10-km buffer in order to capture forested areas under private, state and federal management, as well as shrub and grasslands where fires could ignite and travel into forestland. The final study area covers 357,960 hectares. This landscape is small enough to perform simulations in the LANDIS-II framework with reasonable processing times, while large enough to evaluate the simulated effects of fire and climate interactions across multiple land ownerships and ecosystems.

Temperatures within the study area range from -10°C in winter to over 35°C in summer when most wildfires occur (RAWS Station Data 2018). The majority of precipitation falls as snow or rain in winter and spring, and long precipitation-free periods are common in summer. Topography is complex and characterized by mountains and plateaus descending into steep canyons, with elevation ranging from 500-2,000 m.

Low elevations are characterized by grasslands and sagebrush shrublands giving way to western juniper (*Juniperus occidentalis* Hook.), curl-leaf mountain mahogany (*Cercocarpus ledifolius* Nutt.) and ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson) woodlands further upslope, followed by mixed conifer forests with Douglas-fir (*Pseudotsuga menziezii* [Mirb.] Franco), as well as some grand-fir (*Abies grandis* [Douglas ex D. Don] Lindl.) at mid elevations (Stine et al. 2014; Figure 2.2). The majority of the forestland within the study area is dominated by ponderosa pine and Douglas-fir (Figure 2.3). Mesic sites contain more grand fir, western larch (*Larix occidentalis* Nutt.) and lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. ex S. Watson), with subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) found at high elevations. (Figure 2.2).

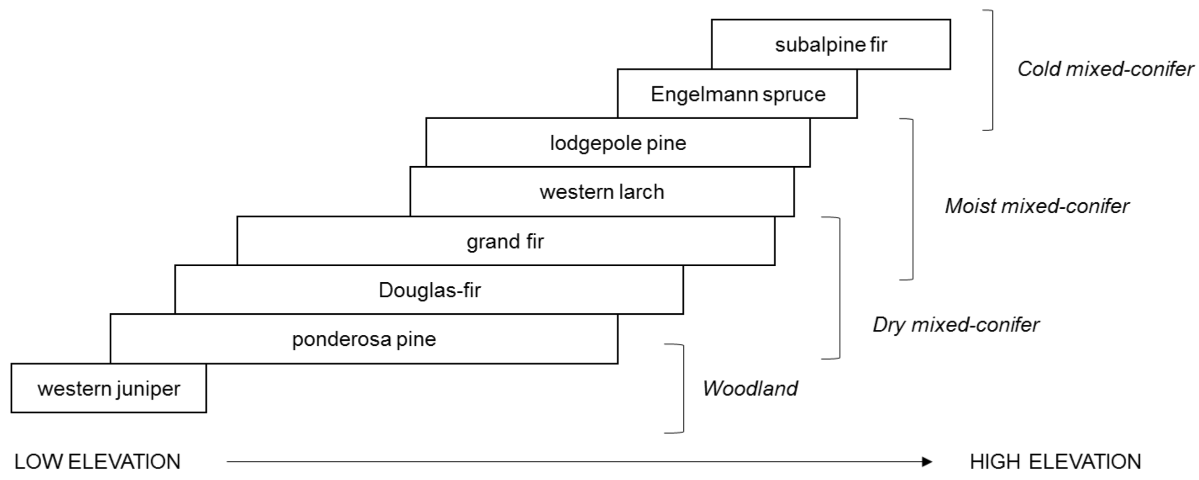

Figure 2.2. Idealized continuum of forest types and dominant tree species from low to high elevations in the Blue Mountains, eastern Oregon. Forest types do not exist in uniform elevation bands across the landscape, but rather intermix at fine scales due to complex topography (e.g., a warm south-facing slope may contain a ponderosa-dominated dry mixed-conifer forest while the cooler adjacent north-facing slope is primarily moist mixed-conifer) (adapted from Stine et al. 2014).

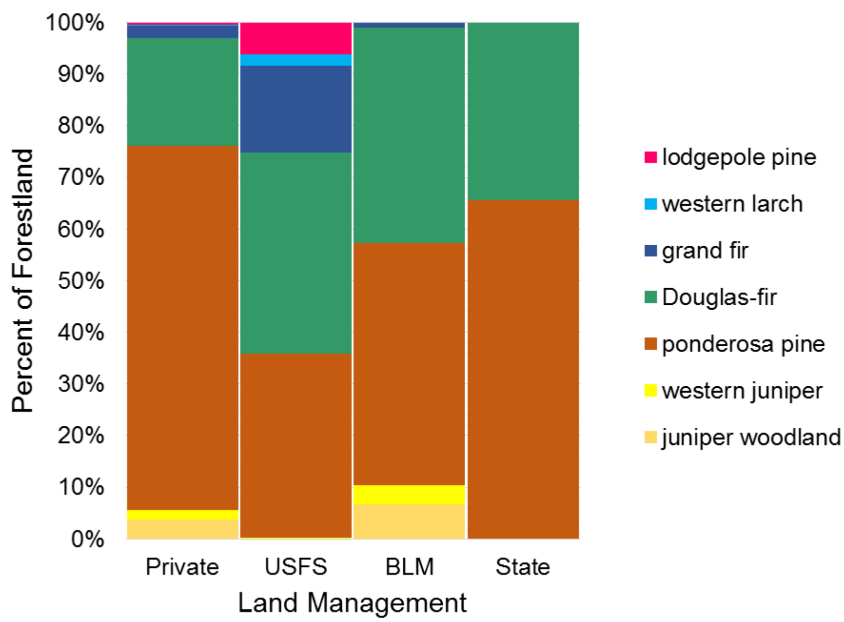

Figure 2.3. Percentage of forestland of each forest type across land management groups within the study area (Ruefenacht et al. 2008), with forest types are defined by Eyre (1980)). Note: Certain forest types, including subalpine fir and Engelmann spruce, are not shown due to small relative percentage of total forestland.

2.3.2 LANDIS-II Forest Landscape Model

I used the LANDIS-II forest landscape model (Scheller et al. 2007) to simulate the effects of climate change and wildfire on conifer forests and characterize projected changes by land ownership. LANDIS-II incorporates ecological processes (e.g. seed dispersal, succession) with disturbances such as wildfire to model forest succession in large landscapes (10^4-10^7 hectares) over decadal to century timescales (Scheller et al. 2007, He 2008). The modeling software framework consists of core libraries as well as independent extensions (Scheller et al. 2007). Within LANDIS-II, the landscape is represented as a grid of interacting cells that contain species-specific cohorts represented by biomass in age classes (Scheller and Mladenoff 2004, Scheller et al. 2007). The model tracks species-age cohorts rather than individual trees to decrease model complexity and limit model run-time and computer memory requirements, enabling spatially explicit modeling over large areas (de Brujn et al. 2014). Within grid cells, species grow, compete, reproduce and die according to user-defined life history parameters, climatic and topographic conditions, and disturbances (Scheller and Mladenoff 2004, Scheller et al. 2007).

There are multiple extensions that can simulate succession and disturbance in LANDIS-II. While there are multiple fire extensions, I used the Dynamic Fuels and Fire System (DFFS v2.1) extension (Sturtevant et al. 2009) so that wildfire ignition and spread was determined dynamically by changing fuel types on the landscape over time (Figure 2.4). I then used the Net Ecosystem Carbon and Nitrogen Succession (NECN v4.2) extension (Scheller et al. 2011) to simulate succession because this extension can calculate probability of establishment dynamically as a function of climate (Figure 2.4; Figure 2.5). The probability of establishment (P_{est}) for each species is internally calculated at an annual time step for each ecoregion and is dependent upon input weather data. Although calculated annually, establishment can only occur following a disturbance or at a succession time step (10 years in this study) (Figure 2.5). P_{est} is based on the minimum of three limiting factors: growing degree days (GDD); drought tolerance; and minimum January temperature. These represent ecoregion-scale limits to species establishment in that the requisite parameters vary by ecoregion (see Section 2.3.3 for details on ecoregions) (NECN User guide v2.1). The user specifies minimum and maximum GDD and a drought tolerance rating for each species during parameterization.

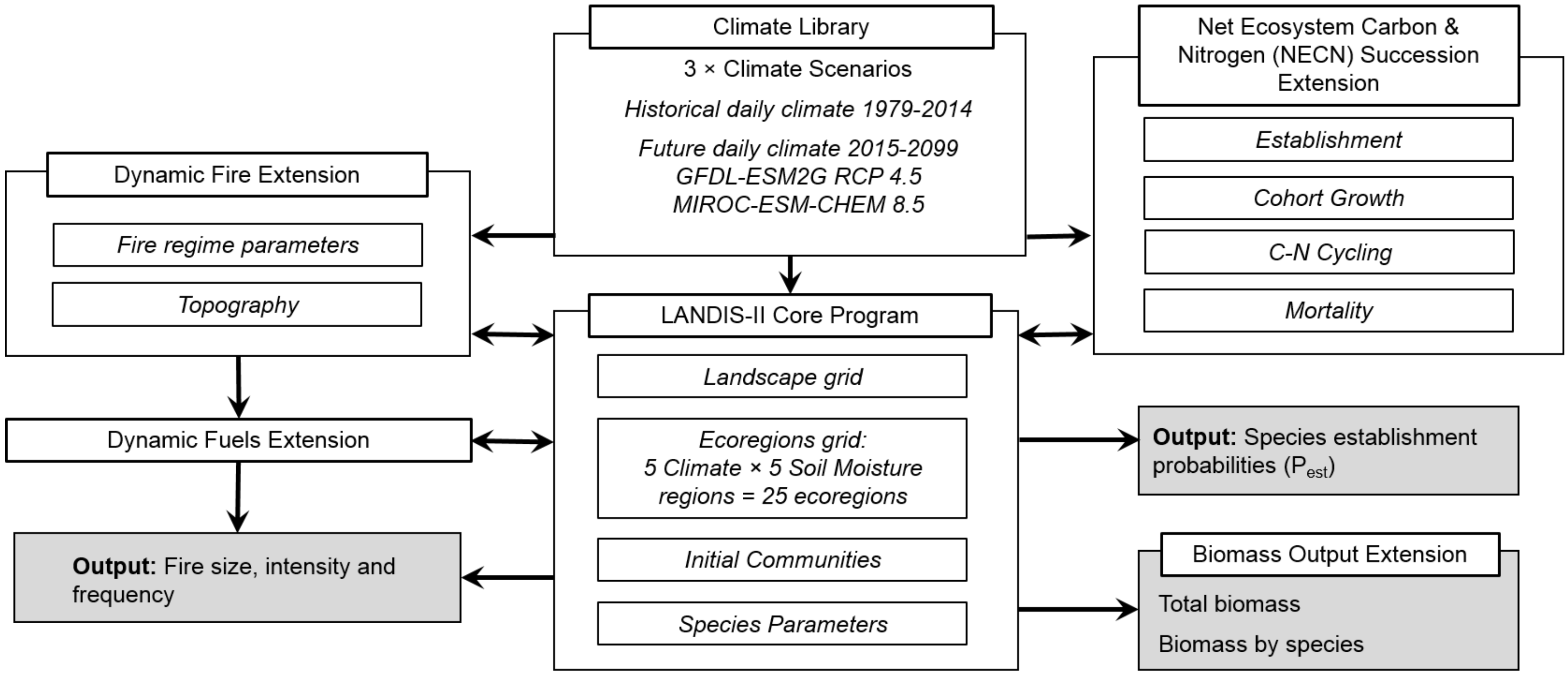

Figure 2.4. LANDIS-II model structure showing relationship between core program, climate library, and extensions used in this study, as well as simulation inputs (white boxes) and outputs (grey boxes).

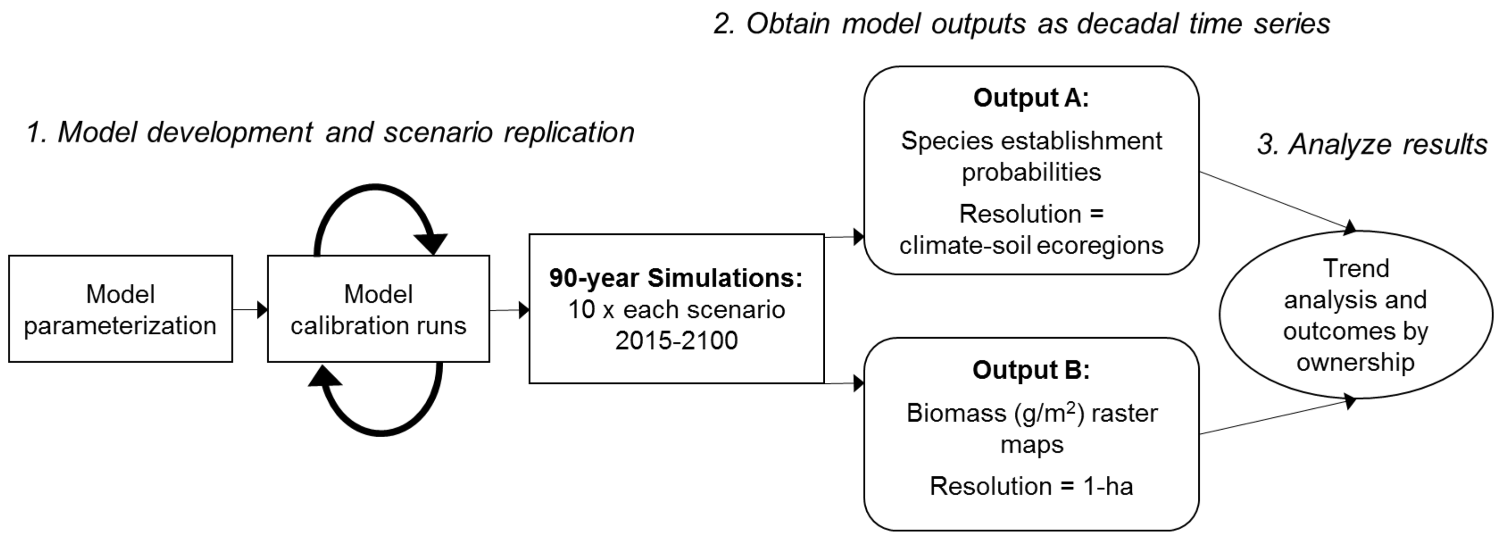

Figure 2.5. LANDIS-II modeling workflow.

2.3.3 Model parameterization

The inputs required for LANDIS-II simulations include several raster maps. First, the model requires a map of environmentally homogenous land types ("ecoregions") based on climate, soil or terrain attributes. I downloaded 30-year (1981 – 2010) normal monthly climate data from the PRISM Climate Group website (www.prism.oregonstate.edu/normals/) at 800-m resolution, including maximum temperature and precipitation for June, July and August, in order to create ecoregions reflecting the annual period of highest moisture stress. I created five climate regions with similar summer temperature and precipitation using isocluster unsupervised classification in ArcMap 10.3, representing a gradient from hot/dry to cool/wet conditions.

To map soil characteristics, I obtained the most recent soil survey data available for the study area comprised of a combination of STATSGO and provisional SSURGO data from the Natural Resource Conservation Service (C. Ringo, pers. comm.). I reclassified available soil water (AWS) at 100-cm depth into five soil regions from low to high soil moisture. I then overlaid the soil moisture regions on the climate regions to create 25 ecoregions with all combinations of climate and soil moisture (Appendix A, Figure A1). I also used these soil data to derive additional soil characteristics at 100-cm depth as required by the NECN succession extension, including clay and sand fraction percentage, drainage class, and other parameters detailed in Appendix A.

LANDIS-II requires an initial ecological community map in which the initial species composition and species age cohorts are specified for each cell of the landscape. I obtained species maps and databases from the Landscape Ecology, Modeling, Mapping and Analysis group at Oregon State University (LEMMA) (Ohmann and Gregory 2002, Ohmann et al. 2011, 2014). I used the 30-m resolution GNN (gradient nearest neighbor) forest structure raster for initial vegetation communities (map region 229) produced for the Northwest Forest Plan. GNN is an imputation modeling technique that combines environmental data with Landsat satellite imagery to produce maps where each grid cell is associated with a Forest Inventory and Analysis (FIA) plot that has the most similar spectral and environmental characteristics (Ohmann and Gregory 2002). The LEMMA forest structure raster contains permanently non-forested cells (e.g. open water) as well as cells that were designated non-forest following initial imputation but have may become forested in future. Where forest growth or wildfire could not occur, such as open water, I reclassified these cells as inactive for the LANDIS-II simulation (Appendix A, Figure A2). While LANDIS-II cannot model complex shrub or grass responses to climate and wildfire variability, I needed to include these plant communities because they commonly carry fire into and between forested areas. Therefore, I used ecotype descriptions in the GNN data to aggregate non-forest cells into non-forest map codes representing five grassland/shrubland functional groups (Appendix A, Figure A2).

I further simplified the number of modeled tree species by analyzing the FIA data linked to the GNN layer and omitting from the model species that occurred on less than 0.5% of the landscape. This resulted in nine final tree species for modeling. While subalpine fir and mountain mahogany are present in low abundances, they are nevertheless ecologically important for conservation. Subalpine fir forests provide food and habitat for numerous high elevation small mammals and birds, including snowshoe hares, red squirrels, pine martens, rodents, and many bird species. Mountain mahogany is an important year-round forage and cover species for ungulates (Johnson and Simon 1987).

In the initial landscape, ponderosa pine covers the majority of forested area, followed by Douglas-fir, grand/white-fir, western juniper, and less common species (Table 2.2). After rounding tree ages to the nearest decade from the FIA-based individual tree database, I used R (R Core Team 2017) to populate an initial communities text file by modifying R scripts from an existing project in the Blue Mountains (Cassell 2018).

Table 2.2. Percent of study area grid cells initially occupied by each species included in the simulation model. **Abies grandis* and *Abies concolor* are known to hybridize with each other throughout central Oregon , and are therefore considered a single species in forest inventories and this analysis (Ott et al. 2015).

Scientific Name	Common Name	Percent of initial landscape occupied
*Abies grandis/concolor**	grand fir / white fir	21%
Abies lasiocarpa	subalpine fir	1%
Cercocarpus ledifolius	curl-leaf mountain mahogany	1%
Juniperus occidentalis	western juniper	18%
Larix occidentalis	western larch	17%
Picea engelmannii	Engelmann spruce	6%
Pinus contorta	lodgepole pine	10%
Pinus ponderosa	ponderosa pine	45%
Pseudotsuga menziesii	Douglas-fir	36%

To simulate species growth, LANDIS-II requires the user to parameterize species-specific life history traits and growth dynamics. The model was recently parameterized and calibrated for mixed-conifer forests in the Blue Mountains 90 km south of my study area on the Malheur National Forest (Cassell 2018). That project in turn modified parameter values and calibration methods from other LANDIS-II projects (Loudermilk et al. 2014, Lucash et al. 2014, Creutzburg et al. 2016, Serra-Diaz et al. 2018). Therefore, I initiated parameterization using existing species life history and growth parameters and calibrated values specifically for my study area (Cassell 2018) (Appendix A). For example, LANDIS-II requires the user to parameterize maximum biomass values for each species. To determine initial values, I summed individual tree biomass by species within FIA plots and calculated the 95th percentile total biomass, which I used as the initial maximum biomass parameter for each species. I modified this parameter during model calibration (see section 2.3.5).

2.3.4 Historical and future climate

I simulated forest succession under three climate scenarios: current climate and two future climate projections. I obtained historical climate data for 1979-2014 from the 4-km resolution daily surface meteorological (METDATA) dataset, which contains interpolated daily precipitation, minimum and maximum temperature, and wind speed values (Abatzoglou 2013a). For future climate scenarios, I selected the MIROC-ESM-CHEM global circulation model (Representative Concentration Pathway (RCP) 8.5), which projects a hot and dry future for the study area, and GFDL-ESM2, which projects a warm and wet future under RCP 4.5. RCP 8.5 reflects business-as-usual increases in global emissions through the 21st century, while in RCP 4.5 emissions peak around 2040 and then decline. I first identified several global climate models with independently determined high validation statistics (Stralberg et al. 2015). Then, I selected these two specific models because I considered them to represent “worst” and “best” case scenarios respectively for trees in my study area in the context of drought stress.

I downloaded projected future daily climate data for these scenarios from the MACAv2-METDATA 4-km resolution dataset through the US Geological Survey Geo Data Portal (Blodgett et al. 2011) (Figure 2.6). The MACAv2-METDATA dataset is derived from a statistical downscaling method for GCM data from the Coupled Model Intercomparison Project 5 (CMIP5) (Taylor et al. 2012) that uses a modification of the Multivariate Adaptive Constructed Analogs (MACA) (Abatzoglou and Brown 2012) with a training dataset (the training data are the METDATA historical observations discussed above (Abatzoglou 2013a)) to remove historical biases and match spatial patterns in climate model output.

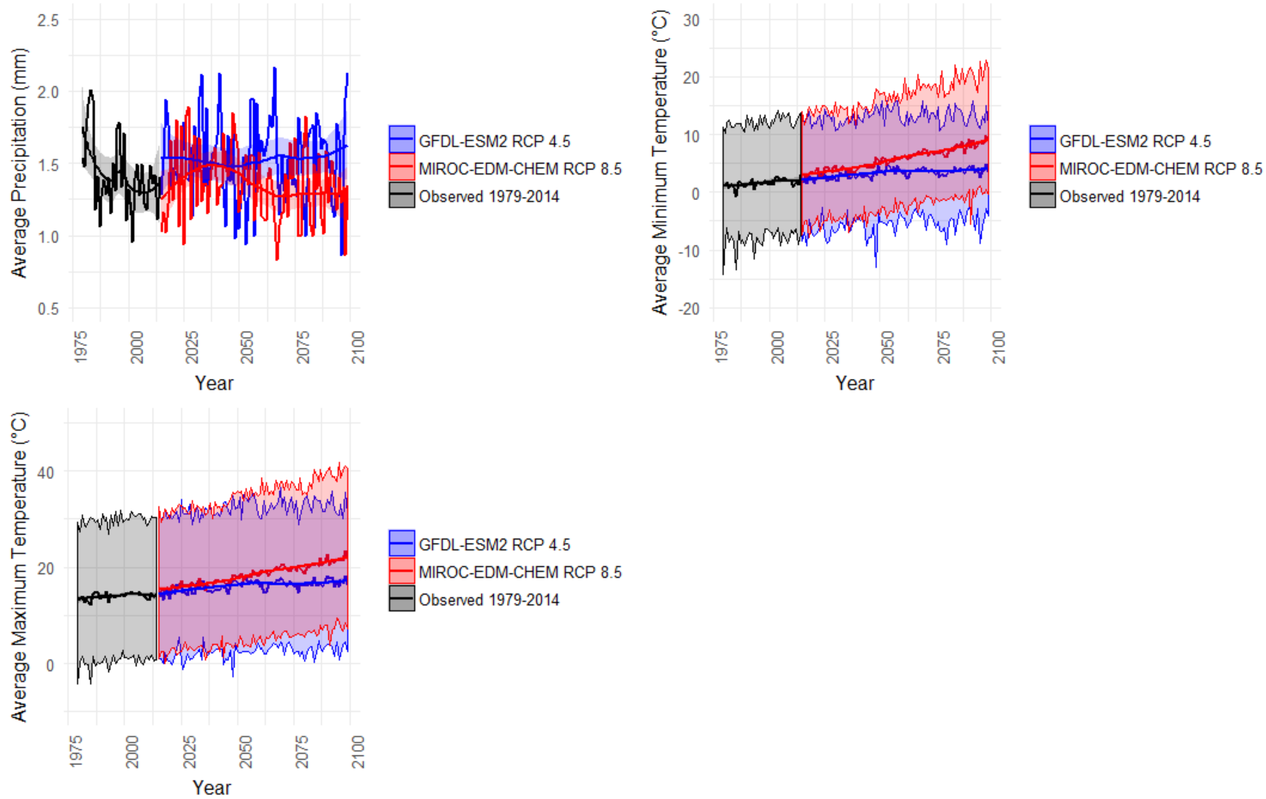

Figure 2.6. Daily historical and projected climate data used in simulation models for the study region, showing average values via a lowess smoother function, in addition to 5th and 95th percentile values. (A) Average precipitation; (B) Average minimum temperature (°C); (C) Average maximum temperature (°C). In each panel, historical observed data (1979-2014) is followed by two climate change scenarios (2015-2100). GFDL-ESM2 projects a "warm/wet" future under RCP 4.5, while MIROC-EDM-CHEM projects a "hot/dry" future under RCP 8.5.

2.3.5 Model calibration

I calibrated the model by changing model parameters individually to align model output with empirical data. I initially calibrated the model using a single cell from the landscape, running the NECN extension in calibrate mode for 100 years using randomized historical climate data. Calibrate mode produces detailed monthly output from the model on carbon and nitrogen pools and fluxes, biomass, and other variables. I calibrated both wet and dry nitrogen deposition to align with empirical observations from the nearest National Atmospheric Deposition Program

(NADP) collection site, located at Starkey Experimental Forest (NADP 2018), as well as CASTNET collection sites (Owyhee and Nez Perce Tribe stations) (EPA 2018).

The model "spins up" by growing initial communities from their establishment based on growth parameters and historical climate, so calibration involves comparing biomass values at "year 0" following spin-up with actual biomass values observed on the landscape to determine if the model is growing trees in a way that reflects reality. I calibrated aboveground biomass (AGB) in the entire study area by comparing the biomass of each species at year 0 with species biomass values from the GNN data. Proceeding one species at a time, I ran simulations using historical climate data and wildfire for 1 year to look at spin-up biomass values at year 0. I modified species-specific parameters for Maximum Biomass and Maximum Monthly Aboveground Net Primary Productivity (ANPP) until simulated average biomass values were highly correlated with empirical data (Appendix A, Figure A3, $R^2 = 0.98$). While it is challenging to independently validate model outputs because FIA data is essentially the only landscape-level data on forest biomass, I also compared model outputs to carbon maps developed by combining FIA data, satellite imagery, and topographic data using methods different from those used in GNN imputation (Wilson et al. 2013). Under the assumption that aboveground live carbon accounts for approximately 50% of AGB (IPCC 2006), my simulated biomass values were within a similar range.

2.3.6 Modeling wildfire behavior

I used the Dynamic Fire and Fuels System (DFFS) extension (Sturtevant et al. 2009) to simulate wildfire. The DFFS requires a spatially explicit map of contemporary fire regime units (FRUs; representing fire rotation periods, e.g. 20, 100 years) in which cells experience similar fire frequency and area burned over time. At each time step in the simulation, the DFFS randomly draws the number of fire ignitions from a Poisson distribution parameterized by the user. For each ignition, the fire module randomly selects a cell in the landscape and evaluates whether the ignition becomes a fire initiation using parameters for the fuel type classification of that cell (Sturtevant et al. 2009). Fuel types are user-defined species and cohort age groups. I used relevant fuel classes from existing studies in the region (Cassell 2018) based on the Canadian Forest Fire Prediction System classifications (Hirsch 1996) (Appendix A).

I parameterized the DFFS using historical fire dynamics in the study area. I downloaded fire spatial data from the Monitoring Trends in Burn Severity database, which uses Landsat satellite imagery to summarize wildfire size and severity information for fires occurring 1984 to present. I considered using fire size distributions for either 1984-2015 or only 2000-2015, because large wildfires in the Blue Mountains have increased in frequency (Hamilton et al. 2016). However, within this small study area, the two datasets did not differ significantly in distributions of fire size or frequency, so I used the full dataset. From 1984-2015, 33 fires were recorded in the study area with a mean size of 1,617 ha (range = 0.5 ha - 21,756 ha ; sd =4,392 ha). Most fires burned in June, July or August. I adjusted the DFFS parameters, including mean, standard deviation, and maximum fire size and ran simulations for 35 years, fine-tuning parameters individually until simulated fire size and frequency did not differ significantly from log-distributed historical data (Appendix A, Figure A4) (Wilcoxon rank sum test, $p > 0.05$). I then parameterized a “High Fire Regime” scenario, in which I increased average fire size by 50% to simulate higher fire activity under climate change. The resulting distribution of fire sizes differed significantly from both the Contemporary Fire Regime scenario and the observed historical data (Wilcoxon rank sum test, $p < 0.05$) (Appendix A, Figure A4). Finally, the DFFS also requires grids of slope and aspect in the study area, which I derived from a 10-m resolution Digital Elevation Model (DEM) of Oregon.

2.3.7 Data analysis

LANDIS-II extensions produce output tables and maps whose type and frequency can be specified by the user. In addition to outputs from NECN and DFFS, I used the Biomass Output extension to summarize biomass changes by species and year (Scheller and Mladenoff 2004). I also developed a Future Probability of Establishment Index (FP_{est} Index) to summarize important ownership-specific trends in P_{est} over the 21st century. This metric essentially provides an indication of “who should be most concerned about what” in terms of climate change impacts.

To develop this index for each tree species I used a Mann-Kendall test to detect significant positive or negative trends in P_{est} within each of the five climate regions (climate region 1 through 5, which encompass the 25 climate-soil ecoregions). If I detected a significant trend for a species within a climate region ($p < 0.05$), I weighted the τ value of the Mann-Kendall test (τ is a measure of the strength of the trend) by multiplying it by the percentage of the land ownership type located within that climate region. I then summed the area-weighted τ values for all climate regions, resulting in the FP_{est} Index for species in each land ownership. High FP_{est} Index values represent significant changes in species establishment probabilities over large proportions of a given land ownership. FP_{est} Index values of 0 indicate no significant positive or negative trend. I also quantified changes in total biomass by species, using Mann-Kendall tests to detect significant trends through time. The Biomass Output extension produces rasters of biomass values each decade. I assessed overall changes in landscape-level abundance using cell-by-cell comparisons of species presence/absence between biomass rasters at year 90 and year 0.

2.4 Results

2.4.1 Changes in probability of establishment

Under the “hot-dry” RCP 8.5 scenario, establishment probabilities declined for all species except ponderosa pine (Mann-Kendall tests, $p < 0.05$) (Figure 2.7). Under this scenario, P_{est} declined for ponderosa pine on State and BLM lands reflecting their location within the warmest climate regions at low elevations. However, P_{est} for ponderosa pine increased on private and USFS lands (Figure 2.8). FP_{est} Index values indicated widespread potential establishment declines for mountain mahogany and lodgepole pine across all land ownerships. Only private and USFS lands were projected to experience declines in subalpine fir establishment, and may also experience widespread declines in Engelmann spruce and grand fir/white fir establishment.

Under RCP 4.5, P_{est} for most species still declined, though less dramatically and with higher variability over time. P_{est} for both Douglas-fir and ponderosa pine increased (Figure 2.7). FP_{est} values were strongly positive for Douglas-fir and ponderosa pine across all land ownerships, indicating significant projected increases in probability of establishment over large proportions of each ownership (Appendix A, Figure A5). In contrast, probability of establishment declined for grand fir/white fir, subalpine fir, western larch, lodgepole pine, and Engelmann spruce on all land ownerships under this scenario (Appendix A, Figure A3). While declining on State and BLM lands, the probability of establishment for juniper increased on USFS lands. Under contemporary climate, P_{est} stayed relatively constant or increased slightly across species (Figure 2.7; Appendix A, Figure A6).

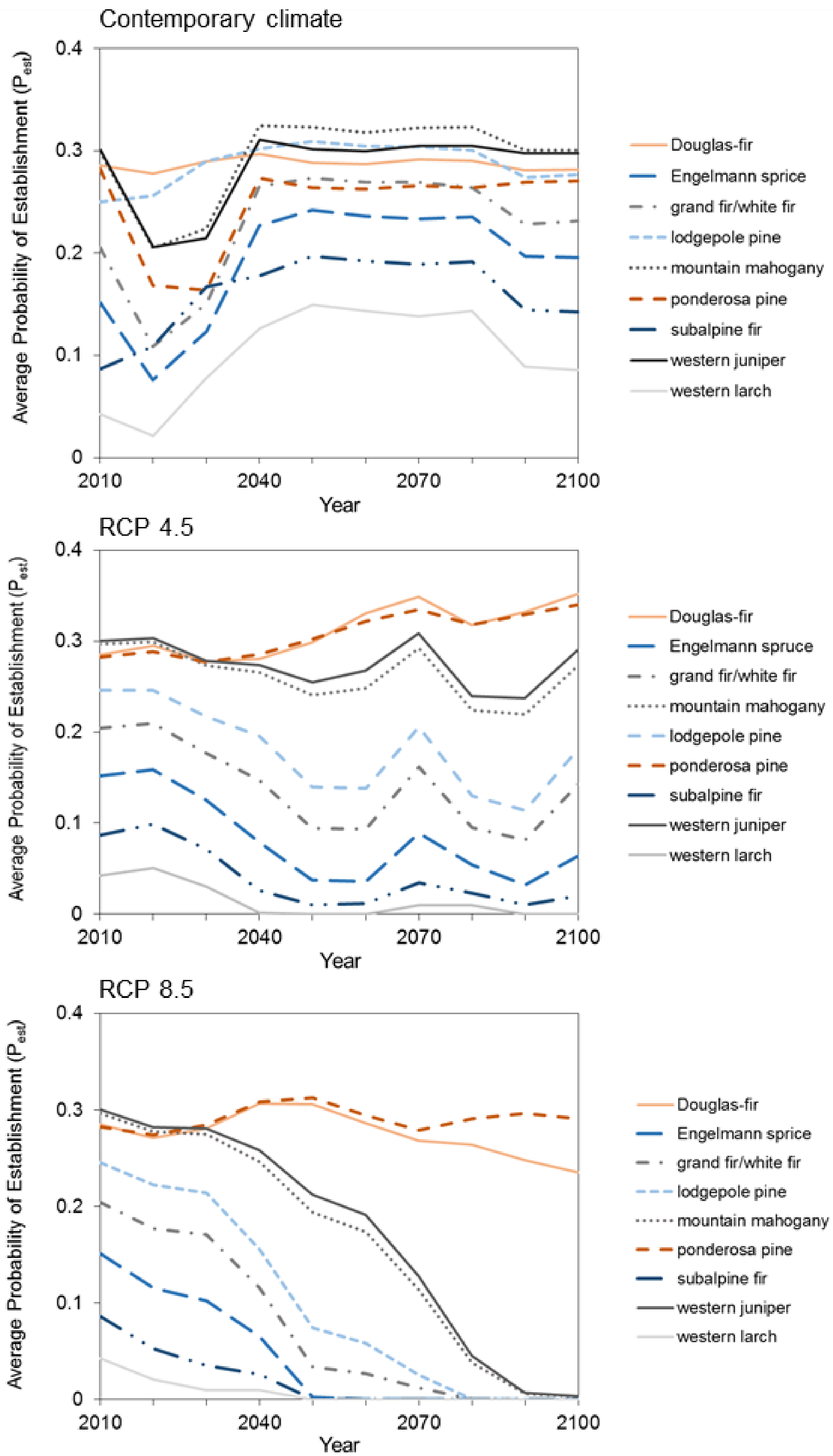

Figure 2.7. Simulated species establishment probabilities (P_{est}) over time, averaged across ecoregions, under contemporary climate and RCP 4.5 and 8.5 climate change scenarios.

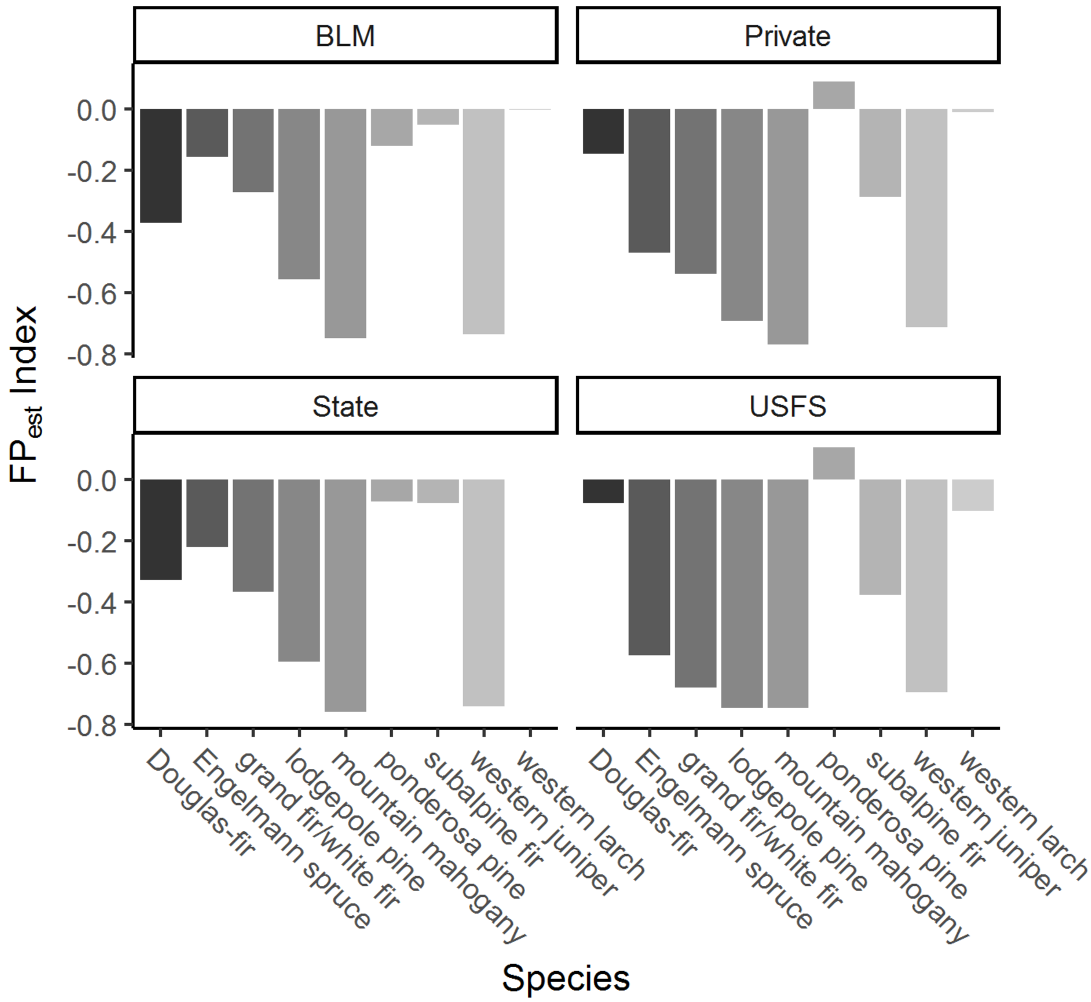

Figure 2.8. Significant changes in species establishment probabilities for 2015-2100 under RCP 8.5 by land ownership, represented by the Future Probability of Establishment Index (FP_{est} Index). The magnitude of the FP_{est} Index reflects significant projected changes (negative values = decline; positive = increase) in species establishment probabilities and the proportion of the landowner group's land area in which the changes are projected to occur (see Appendix A for RCP 4.5 and Contemporary Climate).

2.4.2 Changes in species biomass and abundance

In general, over the course of the simulation period, climate change resulted in potential growth benefits for drought-tolerant tree species and negative impacts for some conifer species adapted to cool-moist conditions. Total biomass increased for most species across most scenarios, though variation in biomass projections was much higher under the high fire scenarios. Douglas-fir, grand fir/white fir, and ponderosa pine increased or maintained total biomass and abundance across all six climate/fire scenarios (Figure 2.9; Figure 2.10; Table 2.3). These three species also experienced the largest increases in total biomass under climate change scenarios compared with contemporary climate. Under both Contemporary and High Fire scenarios and RCP 8.5, subalpine fir and western larch declined in total biomass, while Engelmann spruce experienced declines under High Fire activity regardless of climate scenario. Most species increased their abundance (measured as species extent across the landscape) from 2015 to 2100 (Table 2.3) (Mann-Kendall tests, $p < 0.05$), though increases were generally lower under the High Fire Regime × climate change scenarios. Engelmann spruce, lodgepole pine, western juniper, and western larch declined in extent under the High Fire Regime × RCP 8.5 scenario.

Summing total biomass change 2015-2100 across all tree species reveals greater increases under the Contemporary Fire scenario than High Fire activity, while the three climate scenarios did not produce significant variation in total landscape biomass change (Figure 2.11). Private lands experienced the greatest average increases in biomass across all scenarios (Figure 2.12). USFS and State lands were projected to experience changes in total biomass of similar magnitudes, though biomass projections on USFS lands showed greater variation across scenario replicates. BLM lands experienced the lowest increases in biomass (Figure 2.12).

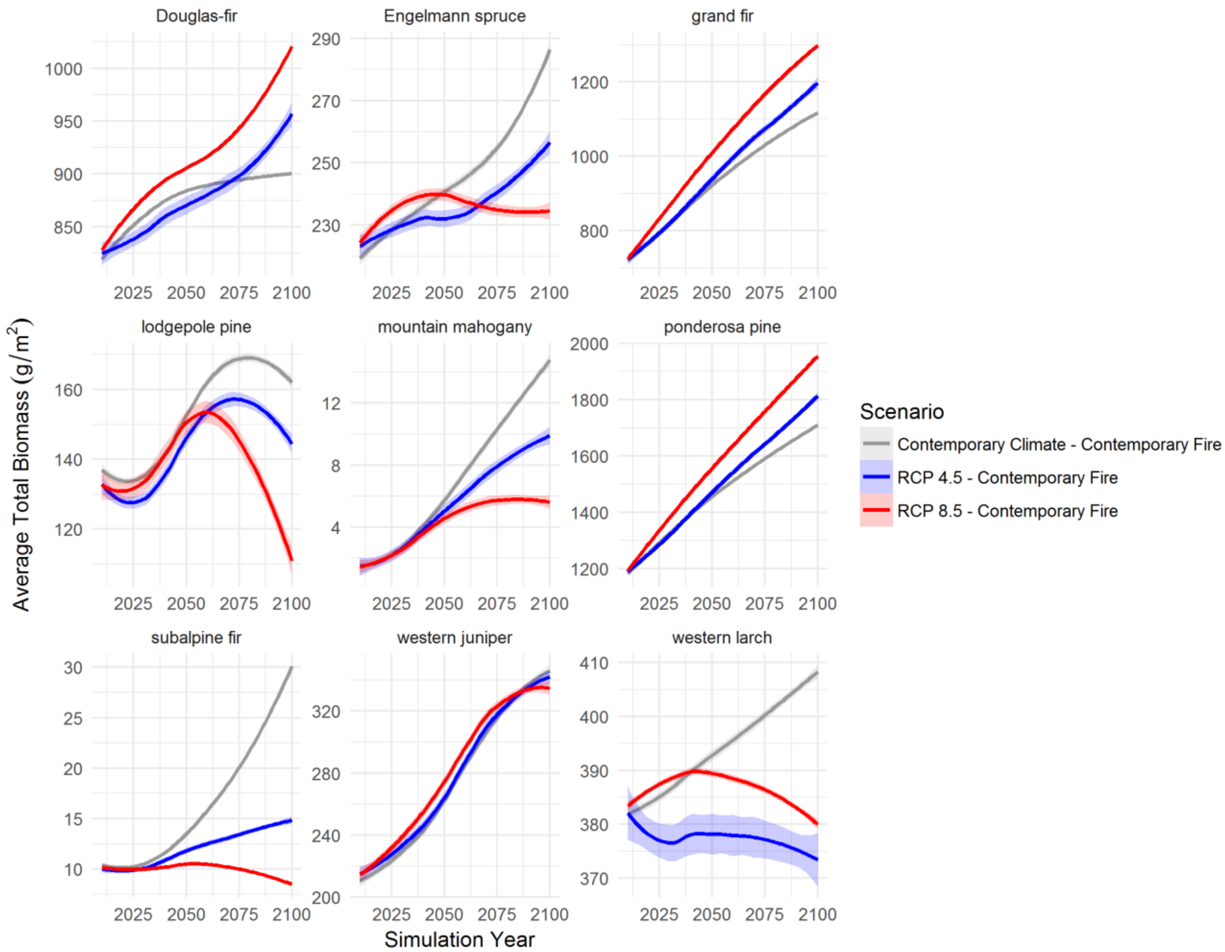

Figure 2.9. Average total biomass (g/m^2) of each species over time under three climate scenarios and the Contemporary Fire scenario.

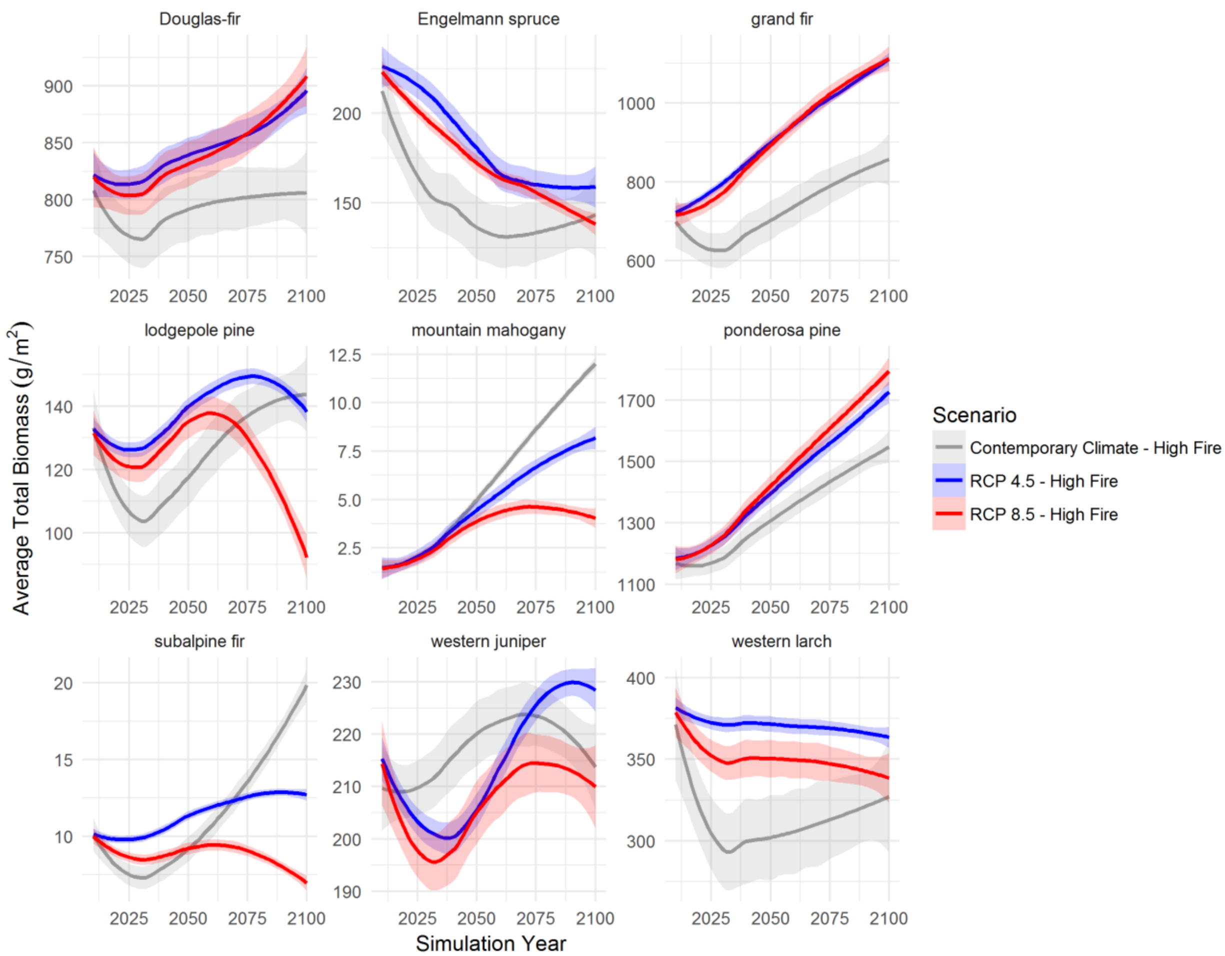

Figure 2.10. Average total biomass (g/m^2) of each species over time under three climate scenarios and the High Fire scenario.

Table 2.3. Average changes in species extent as a percentage of the study area under each climate/fire scenario. Reflecting biomass trends in Figures 2.7 and 2.8, white boxes indicate significant positive projected increase in average total biomass (g/m^2), light grey indicates no significant trend, and dark grey indicates significant projected declines 2015-2100 (Mann-Kendall tests, $p < 0.05$). Note: A species may have a positive increase in landscape extent, but significant decline in total biomass, if biomass per cell (i.e. biomass density) declines (or vice versa).

Species	Contemporary Fire Regime			High Fire Regime		
	Contemporary climate	RCP 4.5	RCP 8.5	Contemporary climate	RCP 4.5	RCP 8.5
Douglas-fir	+18%	+19%	+17%	+13%	+15%	+11%
Engelmann spruce	+9%	+3%	+1.5%	+4%	+1%	-2%
grand fir/white fir	+18%	+10%	+4%	+12%	+8%	+0.5%
lodgepole pine	+4%	+3%	+1%	+4%	+1%	-1%
mountain mahogany	+2.5%	+2%	+1%	+2%	+1%	+0.5%
ponderosa pine	+20%	+21%	+20%	+17%	+19%	+16%
subalpine fir	+4%	+2%	+1%	+2%	+0.5%	0%
western juniper	+14%	+13%	+7.5%	+6%	+4%	-6%
western larch	+2%	0%	0%	+2%	-1.5%	-3%

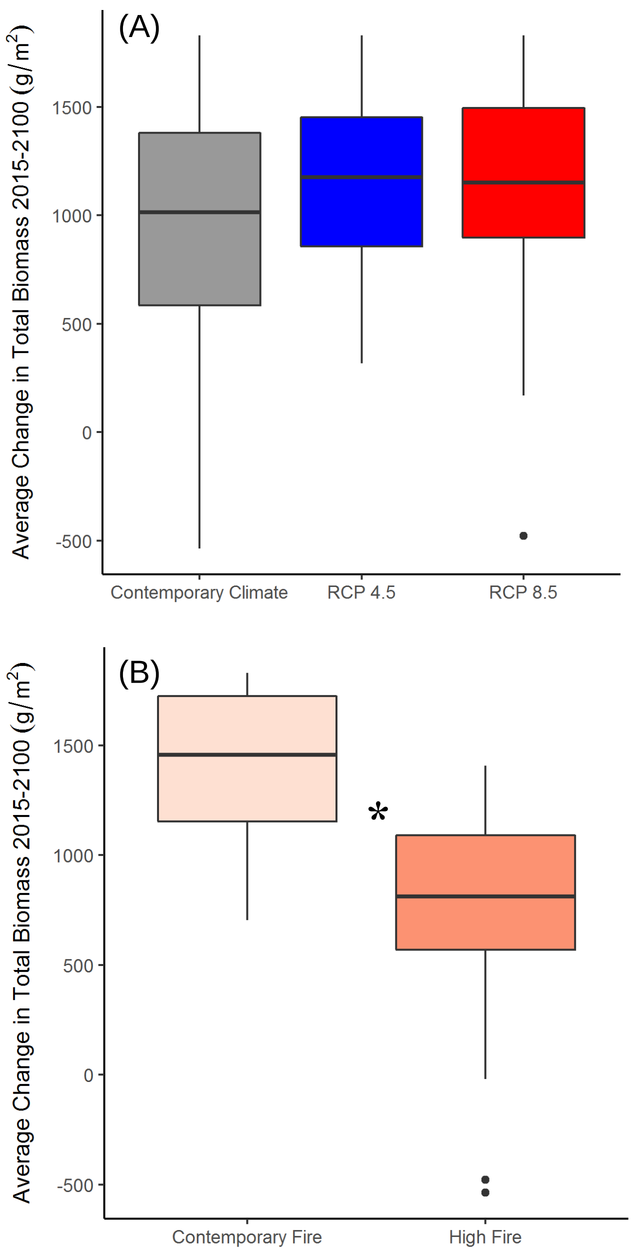

Figure 2.11. Average change in total tree biomass across the entire study area across all simulations. Panel A) shows no significant difference in biomass change between three climate scenarios; and B) significant differences * between two wildfire scenarios. ($t = 11.4$, $p < 0.05$).

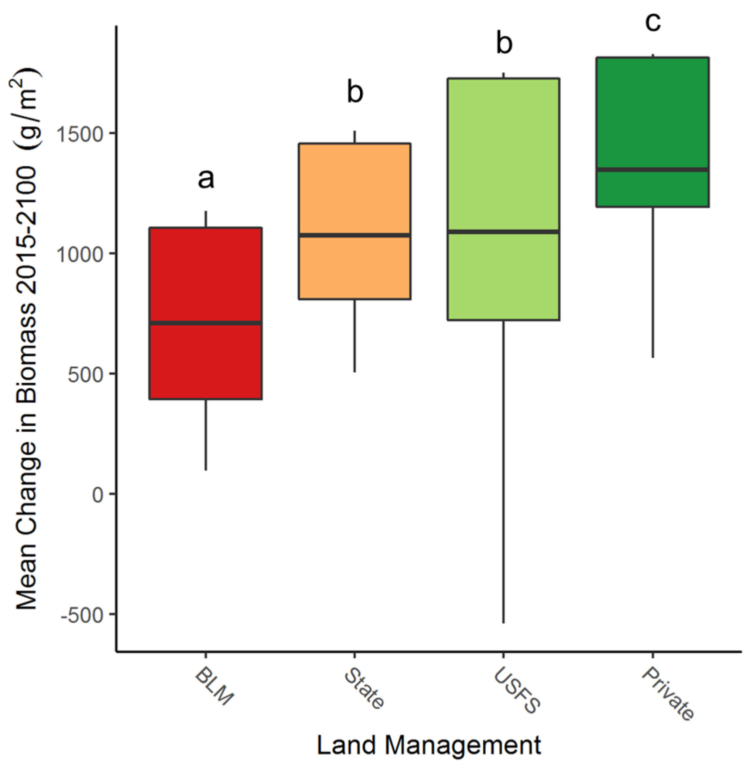

Figure 2.12. Mean change in biomass across all scenario replicates and land management classes. Different letters indicate significantly different group means (Tukey HSD, $p < 0.05$).

2.5 Discussion

Changing climate and fire regimes demand region-specific information on priorities for adaptive forest management. In this study, I evaluated the effects of a range of future climate and wildfire scenarios on tree species in eastern Oregon, specifically assessing projected effects by land ownership. In line with other modeling exercises in the inland northwest US, my simulations suggest montane and subalpine conifer species including subalpine fir, lodgepole pine, Engelmann spruce and western larch may experience declines in establishment and/or aboveground biomass under climate change (Halofsky et al. 2014, Campbell and Shinneman

2017, Cassell 2018, Kim et al. 2018). Declines in cold mixed-conifer forest will reduce habitat for a variety of local wildlife, including snowshoe hare, pine marten and Clark's nutcracker.

In contrast, I found drought-tolerant and fire-resistant ponderosa pine and Douglas-fir may benefit from climate change and high wildfire activity, increasing in biomass and expanding across 10-20% of the study area by 2100. I projected increasing establishment probabilities for ponderosa pine and Douglas-fir overall under climate change, despite establishment declines in the hottest, driest ecoregions in the study area at low elevations. Ponderosa pine and Douglas-fir are currently energy-limited on cooler sites at high elevations, and under warming conditions will likely expand into high elevation areas. Taken together, these projected changes may be beneficial from a timber production perspective.

My results also emphasize the critical role wildfire plays in altering landscape-level successional trajectories and regional forest structure. High wildfire activity resulted in more variation in biomass projections than different climate scenarios, indicating that the timing and location of large wildfires will likely have greater implications for forest change in the 21st century than climate warming alone. Other forest landscape modelling studies in central and eastern Oregon project more frequent, more extensive, and higher severity fire activity under climate change (Serra-Diaz et al. 2018, Cassell 2018). Such increased fire activity may accelerate declines in the extent of fire-intolerant montane and subalpine tree species, and facilitate the expansion of fire-tolerant species such as ponderosa pine. Notably, in my simulations Engelmann spruce was far more sensitive to increases in wildfire activity than increasing temperatures.

2.5.1 Ownership-specific implications for climate-adaptive forest management

My findings suggest different management priorities across different land ownerships within the study area. From a silvicultural perspective, managers on state and BLM lands as well as private lands at low elevations may experience the most negative impacts. Under high rates of warming and drying they may see establishment declines in ponderosa pine and Douglas-fir, the two dominant tree species on these lands. In contrast, the US Forest Service manages very little land in the hottest, driest parts of the landscape, but may face management challenges in their high elevation forest types where cold-adapted species will decline under warming conditions. These contrasting challenges mean these land managers will have different priorities for climate-adaptive forest management.

Climate-adaptive management actions generally fall under three approaches: resistance, resilience, and transition (Millar et al. 2007, Ontl et al. 2018). Forests may be managed to resist changes, such as preserving mature stands of climate-vulnerable species so that they will persist via inertia as the climate warms. Alternatively, managers can take actions to enhance the forest's ability to recover to its original state following disturbance. Lastly, where ecosystem change is inevitable, managers can facilitate ecosystem transitions by favoring species that may be better adapted to future conditions. The utility of these three approaches varies among forest types.

In ponderosa pine woodlands and dry mixed-conifer forests in the Blue Mountains, where low-severity fires were historically frequent, a primary management goal now and in the future is maintaining resilience. Thinning and prescribed fire aim to reduce the risk of high-severity fires that dramatically alter the ecosystem (Chmura et al. 2011, Hessburg et al. 2015, Halofsky et al. 2016a). Additional climate-smart actions include thinning stands to wider-than-traditional spacing in anticipation of higher moisture stress and more frequent fire activity (Halofsky et al.

2016b, 2018a), reducing planting density, and planting diverse genotypes or species that may be better adapted to future conditions (Chmura et al. 2011, Nolan et al. 2018). Additional actions include favoring certain species depending on aspect and strategically thinning around legacy trees (i.e., old, large trees that are more likely to survive disturbance) (Halofsky et al. 2016b). All of these climate-smart management actions should be priorities for BLM and State forest managers within this study area, since dry forest types constitute the majority of their land holdings. These actions should also be priorities for USFS and private landowners in the dry forests they manage.

In contrast, in cool and moist forest types where wildfires were historically driven by climate, weather, and lightning ignition rates rather than fuels (Schoennagel et al. 2004, Halofsky et al. 2018b), fuels treatments are likely not an effective adaptation approach. In forests with historically infrequent stand-replacing fire regimes, one adaptation strategy may actually be to continue fire suppression where possible and where it is potentially beneficial for wildlife, carbon sequestration, municipal watershed health, and for maintaining refugia for vulnerable tree species (such as the Engelmann spruce and subalpine fir identified in this study). Fire suppression in these forest types will not change forest structure substantially from its natural range of variation (Halofsky et al. 2018b), and therefore may have less negative impacts than in dry forest types.

Other adaptation options in stand-replacing regime forests include maintaining species diversity and structural diversity (Halofsky et al. 2018b). High species diversity can hedge uncertainties regarding changing establishment probabilities and growth responses. Managing for structural diversity involves promoting diverse successional stages, rather than focusing on even-age production stands. Structural diversity may promote resilience to climate-related stressors such as insects and disease (Raymond et al. 2014). Finally, connectivity between vulnerable forest types should be maintained by creating management agreements across National Forest and adjacent private forestlands as an all lands approach to ecosystem conservation.

Obviously, some of these adaptation actions are easier to implement than others. Selective fire-suppression is challenging, given that fires beginning in dry conifer forest (where they are desirable) naturally may burn upslope into other forest types. Additionally, maintaining structural or compositional diversity may conflict with some private landowners' timber production goals, for example, if they maintain stands of a fast-growing single species such as western larch. Private landowners and USFS staff need to consider the above adaptation actions given the large areas of upper montane and subalpine forests they own and manage. Both landowner groups will need to decide how to respond to the conversion of cool/moist mixed-conifer forest to dry mixed-conifer or ponderosa pine forest. Some managers may see this conversion as beneficial due to ponderosa pine and Douglas-fir timber values. However, the Forest Service has a multiple use management mandate, and they need to prioritize resistance, resilience, or transition at fine scales depending on specific ecological and social goals.

A low-risk climate-smart management strategy for all land managers would be establishing management experiments following stand-replacing disturbances or harvests. Owners and managers could allow one portion of an area to regenerate naturally, plant one portion with a novel mix of genotypes and species at varying densities, and plant another using traditional species and densities (Halofsky et al. 2018b). As climate and wildfire regimes continue to shift, the results of such experiments will inform future decision making. Modeling studies like the forest landscape model presented here can also be used to "experiment" with different potential adaptation actions, including understanding the effects of "climate suitable planting" (CSP) on future forest ecosystems. Also referred to as assisted migration, CSP involves planting tree species beyond their current range in places where the climate is expected to become suitable in the coming decades. LANDIS-II has been used in other regions to project the effects of CSP under multiple climate change scenarios (Duveneck and Scheller 2015, Hof et al. 2017). However, as with all forest management activities, initiatives such as CSP require resources.

All climate-smart forest management actions require planning, funding, and monitoring for implementation to be successful. Resources are limited for federal agencies and private landowners alike, and landowners and managers will need to prioritize actions they deem most important given the forest types they manage, their current structure and composition, and their management goals. For climate-adaptive management on federal lands, strong social networks and collaborative groups composed of members of the public, scientists, and managers, can foster the communication and engagement to secure the necessary social license to implement actions at broad scales (Halofsky et al. 2016b). To support private forest owners, local stewardship and consulting foresters need training on integrating climate change into forest stewardship plans. Additionally, many private forest owners in eastern Oregon lack the funds, labor pool, and equipment to conduct the fuel treatments they would like to complete on their lands (Boag et al. 2018). Grants, cost-share programs, and collaborative fuel treatment efforts between landowners and across ownerships may be avenues to reduce these barriers. However, a major remaining barrier to engaging private landowners in the Blue Mountains on climate-smart forest management is high levels of politically-based skepticism regarding the existence of anthropogenic climate change (Hamilton et al. 2016). While wildfire mitigation can be used to justify many adaptation actions, I would argue climate change must be invoked to engage private landowners on actions such as climate suitable planting or protecting high elevation refugia. This barrier may be difficult to overcome without a larger cultural shift in views on climate change.

2.5.2 Study limitations

I parameterized and calibrated the dynamic forest landscape model used in this study with the best available data; however, as with every ecological model, it contains many sources of uncertainty. The specificity of the model is limited by the availability and quality of empirical data from this region for parameterization and calibration. The number and complexity of ecological processes included in the model is also constrained by time and computational resources. This study omitted some species that exist in the study area but cover too small an area to be meaningfully modelled using this framework. I recognize that these species, such as trembling aspen (*Populus tremuloides*), may nevertheless be conservation priorities for landowners and managers.

Additionally, this model does not incorporate dynamic management processes such as harvesting or prescribed burning because it was beyond the scope of this study, though extensions exist in the LANDIS-II environment to incorporate these processes. Future research could investigate how different management scenarios affect wildfire and carbon sequestration dynamics in a changing climate (Creutzburg et al. 2017). Similarly, I did not dynamically model mortality due to wind, disease or insect disturbances, the behavior of which may also change under climate change (Scheller et al. 2018) and interact with other disturbances, such as wildfire, to alter forest structure in non-linear ways (Lucash et al. 2018). However, I did incorporate estimates of the contribution of these processes to mortality in the model’s base disturbance and mortality parameters.

The newest version of NECN (v5) was released after this model was parameterized. It eliminates ecoregions and calculates probability of establishment on a cell-by-cell basis. Revising all of the inputs to operate in the new version would allow changes in species establishment probabilities by land ownership to be analyzed directly, as opposed to by ecoregion, as done here. Finally, the stochastic nature of this model leads to a range of outcomes stemming from each combination of scenarios, which is why I averaged results from 10 replicates of each experimental condition. Computational constraints limited the feasibility of running more replicates per scenario.

2.6 Conclusions

Since 1975, eastern Oregon has warmed by more than 1.5°C, twice the global rate of change over this period (Hamilton et al. 2016). Unless global emissions are dramatically curtailed, this trend will continue (IPCC 2018). This study adds to existing projections that upper montane and subalpine tree species in the inland northwest will be vulnerable to climate change impacts and may face extirpation, while ponderosa pine and other drought- and fire-tolerant species may benefit. Such shifts are likely to negatively impact regional biodiversity. My research also supports the theory that larger wildfires increase uncertainty around future forest structure and composition, and will be a primary driver of ecosystem shifts over the 21st century. The implications of these findings should be integrated into ongoing forest management planning and actions.

Firstly, public land managers should collaborate with private forest owners across property boundaries to enhance connectivity between refugia for vulnerable tree species.

Secondly, in cases where wildfire managers have any control over fire behavior, they should communicate with forest ecologists to determine where fire may be most ecologically beneficial or ecologically detrimental at local scales. Third, where climate change may impact key life stages, including reducing seedling establishment of Douglas-fir and ponderosa pine at low elevations, landowners and managers should duplicate and share results from planting experiments with drought-hardy provenances or genotypes. Finally, wildfire activity will likely keep rising in eastern Oregon, and managers should use burned areas to experiment with climate-smart approaches including varying planting density and composition.

As climate and wildfire activity continue to change, private forest owners will require management recommendations and educational resources from state forestry agencies and university extension services that address the broad range of forest types they manage. Owners and managers will need to decide on approaches for specific contexts, including whether to resist change, support resilience, or facilitate inevitable transitions. Ideally, these decisions will be made in the context of the broader forested landscape. An all lands approach to climate change adaptation will create the ecological heterogeneity and redundancy necessary to maintain biodiversity and ecosystem services in forests now and in the coming decades.

CHAPTER III

TOPOGRAPHY AND FIRE LEGACIES DRIVE VARIABLE POST-FIRE JUVENILE CONIFER DENSITIES IN EASTERN OREGON, USA

3.1 Abstract

Increasingly frequent large wildfires in the western US raise questions about the effects of post-fire climate and site-level factors on forest ecosystem resilience. This study presents findings from seedling and sapling surveys conducted across 184 sites 15-21 years post-fire in eastern Oregon's Blue Mountain ecoregion. I found wide variation in conifer seedling and sapling densities across low, medium and high severity burn areas in the eight fires surveyed. One-third of sites had zero seedlings and saplings, while a quarter of sites had densities above 2,000 juvenile trees ha^{-1}, in part due to dense lodgepole pine saplings. The most important variables explaining juvenile conifer presence and absence were heat load, overstory density, and distance to live seed source. Regional drought conditions within the first 3 years post-fire were not one of the most important predictors of juvenile conifer presence after accounting for site-level variables, though drought may interact with topography to reduce regeneration. I rarely recorded seedlings and saplings more than ~100-m from a living seed source, and saplings were more frequently observed at higher elevations for both lodgepole pine and other species. I assessed the adequacy of seedling and sapling densities by comparing observed juvenile conifer densities to local stocking recommendations for specific plant associations. I found densities did not meet minimum stocking recommendations in approximately 35% of sites, primarily in Douglas-fir, ponderosa pine, and grand fir-dominated plant associations. My findings suggest

topography is a key driver of post-fire juvenile conifer densities in eastern Oregon, and that juvenile conifer densities in large, high-severity burn patches on warmer sites may be insufficient to meet local silvicultural guidelines or maintain forest ecosystem function without supplementary replanting. Some of these marginal sites may be susceptible to ecosystem state transitions to shrub or grasslands, while juvenile trees are extremely abundant in high elevation, cool sites. The findings from this study inform post-fire and climate-adapted forest management in the inland Northwest.

3.2 Introduction

The frequency and area of large wildfires in the western US is increasing (Dennison et al. 2014, Williams and Abatzoglou 2016) raising questions about post-fire forest recovery (Buma and Wessman 2013, Savage et al. 2013, Enright et al. 2014, 2015, Harvey et al. 2016). Given the current trajectory of global greenhouse gas emissions, climatologists predict further temperature increases will raise the frequency and severity of drought in the western US and increase the frequency and size of wildfires (Williams and Abatzoglou 2016, Abatzoglou and Williams 2016). Climate-mediated changes in the frequency, size and intensity of disturbances will lead to changes in forest structure and function (Turner 2010, Vose et al. 2012, Enright et al. 2015). Additionally, shifting fire and climate regimes alter forest ecosystem services, including carbon sequestration and the regulation of hydrological processes (Saxe et al. 2016, Hurteau 2017).

In response, an increasing number of studies investigate how topography, burn severity, and climatic factors affect post-fire forest recovery, recognizing that succession dynamics vary regionally. Post-fire forest recovery may be limited or may fail if wildfires are followed by drought (Enright et al. 2015), or if high-severity burn patches are too large to be naturally re-seeded by the nearest surviving trees (e.g. Harvey et al. 2016). Accumulating evidence suggests that in some regions interactions between drought and wildfire may convert forest ecosystems to alternate shrub or grassland states that persist for substantial periods of time (Scheffer et al. 2001, Lenihan et al. 2008, Biggs et al. 2009, Anderson-Teixeira et al. 2013, Enright et al. 2015, Johnstone et al. 2016). In the southwestern US following large high-severity burns, some ponderosa pine forests have transitioned to shrubland or grassland ecosystems (Roccaforte et al. 2012, Savage et al. 2013).

Outside of the southwestern US there is increasing information on the rate and density of conifer regeneration following large stand-replacing fires (Bonnet et al. 2005, Donato et al. 2009, 2016, Buma and Wessman 2013, Harvey et al. 2016, Chambers et al. 2016, Rother and Veblen 2016, Stevens-Rumann et al. 2018). In the Colorado Front Range, surveys indicate limited or no regeneration in large high severity burn patches far from live seed sources (Chambers et al. 2016, Rother and Veblen 2016). In California's Sierra Nevada, stocking densities of ponderosa pine were deemed insufficient in high-severity burn patches (below 200 seedlings ha^{-1}, a regional threshold established by local silviculturists) (Crotteau et al. 2013, Collins and Roller 2013). These studies suggest replanting may be necessary to maintain dry conifer forest cover.

Studies in regions with more cool-moist forest have produced varying results on post-fire regeneration dynamics across forest types. In the northern Rockies declines in post-fire seedling establishment for Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) were observed with increasing post-fire drought severity and distance to live seed source (Harvey et al. 2016). However, no impacts were detected for western larch (*Larix occidentalis*), Douglas-fir (*Pseudotsuga menziesii*) and serotinous lodgepole pine (*Pinus contorta*). In Montana and Idaho, regeneration also declined with distance to seed source and lower pre-fire basal area, however regeneration was abundant overall (Kemp et al. 2016). Research from Montana and Idaho also indicates 75% of high severity burn patches in the region were small enough that seed limitation was not impacting the resilience of forests to large wildfires (Kemp et al. 2016). In Oregon's eastern Cascade Range, a study in a decade-old burn found that seedlings naturally established at high densities above 1,030-m elevation, while all plots below this elevation had no well-established seedlings, likely due to moisture-limitation (Dodson and Root 2013). Finally, a meta-analysis of data from across the western US indicates increasingly unfavorable conditions for regeneration due to rising annual moisture deficits related to climate change (Stevens-Rumann et al. 2018). Low elevation dry forests may be vulnerable to ecosystem state transitions post-fire, while moist forests may shift in structure and composition (Stevens-Rumann et al. 2018). My research builds on the developing post-fire forest recovery literature by assessing the variables influencing natural post-fire conifer regeneration in the Blue Mountains ecoregion of eastern Oregon.

3.2.1 Research aims

The frequency of wildfires in eastern Oregon has increased since the mid-1980s as it has in some other parts of western North America (Hamilton et al. 2016, Abatzoglou and Williams 2016). Oregon experienced its most expensive wildfire season to date in 2017, costing $454 million – triple the average annual cost between 2010 and 2015 (Statesman Journal 2018). Rising fire-fighting costs are reducing funds available to the US Forest Service for other programs, including mitigation and restoration activities (Steelman 2016). These funding constraints enhance the need to understand regional post-fire recovery in order to prioritize areas for post-fire management. This study aims to understand where forests may be resilient to shifting wildfire and climate regimes and are naturally regenerating, where managers should resist ecological state transitions by replanting, and where transitional or transformational approaches – allowing forest to transition to non-forest – may be warranted (Millar et al. 2007). Specifically, I aim to understand which site-level variables are most important in determining post-fire conifer regeneration.

If my results align with those of a recent meta-analysis (Stevens-Rumann et al. 2018), I expect to observe more regeneration in fires that burned in the mid-1990s compared with those that burned at the turn of the 21st century. The late 1990s in eastern Oregon were characterized by cooler, wetter conditions compared to moderate to extreme drought conditions from 2001-2004 (NOAA 2018). I also hypothesize that the Blue Mountains landscape position and topography (elevation and aspect) are likely important determinants of post-fire forest regeneration. Drier sites at low elevations and south-facing slopes will likely have lower seedling densities than those at higher elevations and cooler aspects, and densities are expected to decline with distance to seed source as demonstrated by the numerous studies discussed above.

3.3 Methods

3.3.1 Study area

The Blue Mountains ecoregion (25,275 mi^2 or 65,000 km^2) is comprised of semi-arid shrub and grassland and forested uplands in eastern Oregon, southern Washington and western Idaho. The Blue Mountains have a deep history and culture of ranching, agriculture and logging. The local timber industry collapsed in the early 1990s, severely impacting the socioeconomic fabric of local communities, however several industrial logging companies still operate locally and some non-industrial private forest owners do commercial timber sales (Christoffersen 2005). Over half the land is federally managed in national forests (Wallowa-Whitman, Umatilla, Malheur, and Ochoco) and wilderness areas.

The forested landscape is dominated by dry mixed conifer forests at low elevations comprised primarily of ponderosa pine (*Pinus ponderosa*), as well as Douglas-fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*), giving way to western juniper (*Juniperus occidentalis*) shrubland and grassland on southern aspects and lower elevations. At mid elevations and cooler aspects moist mixed-conifer forests contain more grand fir and western larch (*Larix* *occidentalis*), while cold upland sites are characterized by subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), lodgepole pine (*Pinus contorta*), and whitebark pine (*Pinus albicaulus*) in high alpine areas. Local silviculturists debate levels of serotiny in lodgepole pine in the Blue Mountains, with some adhering to historical observations suggesting no or very little serotiny. More recently, local forest ecologists have recorded serotiny in approximately 20% of Blue Mountain lodgepole stands and report nearly pure stands in some sites following mixed and high severity fires on high elevation plateaus (Agee 1996).

The Blue Mountains are topographically complex. Elevation ranges from 700 to nearly 3,000-m. Steep canyons descend from mountains and windswept plateaus, creating abrupt aspect transitions. Most precipitation falls as snow during the winter or rain in spring and fall, while summers are hot and dry with daily temperatures at lower elevations commonly above 30°C for multiple weeks. The fire season generally runs from the end of June through the end of September, with most wildfires caused by lightning ignitions, though an increasing number due to human ignitions (Palace, M. unpublished data).

3.3.2 Sampling design

As this is the first widespread survey of post-fire conifer regeneration in the Blue Mountains, I was interested in characterizing regeneration across a variety of forest types. However, I constrained sampling to areas mapped as Douglas-fir and ponderosa pine-dominated potential vegetation types (PVTs) to maintain some consistency in plant associations surveyed, though these associations do exist across a wide range of dry, moist, and cold mixed-conifer forest types. I further constrained sampling to upland forest below 1,700-m (5,500 ft) that burned in wildfires in eastern Oregon in two active fire periods, 1996 (Bull; Tower; Summit; Wheeler Point) and 2000/2001 (Bridge Creek; Monument Complex; Milepost 244; and Hash Rock) (Figure 3.1). The upper elevational limit aimed to omit subalpine communities. The summer of 1996 was followed by three years of normal moisture based on Palmer Drought Severity Index (PDSI) values (0.3 SD above 1970-2000 average annual PDSI), while the summer of 2000 was followed by three years of moderate drought conditions (1.3 SD below 1970-2000 average annual PDSI) (NOAA 2018), with corresponding variation in the Standardized Precipitation Evapotranspiration Index (SPEI) (Figure 3.1). SPEI is a drought index that accounts for the effect of temperature on moisture availability by incorporating both temperature and precipitation data (Vicente-Serrano et al. 2010). I downloaded precalculated SPEI values for post-fire growing seasons (April 1 – September 30) at 4-km resolution (Abatzoglou 2013b) from the Climate Engine, available online: https://clim-engine.appspot.com/ (Huntington et al. 2017).

Sites were surveyed between June and August in 2016 and 2017. Selecting sites that burned at least 15 years previously helped ensure early seedling establishment and mortality has stabilized (Newton et al. 2006). All candidate fires had > 300 ha of medium or high burn severity area based on wildfire incidence, size, and severity data from the Monitoring Trends in Burn Severity Program (MTBS 2017). Sites were located on public land, including US Forest Service, Bureau of Land Management, and Oregon Department of Fish and Wildlife land, as well as in Wilderness Areas.

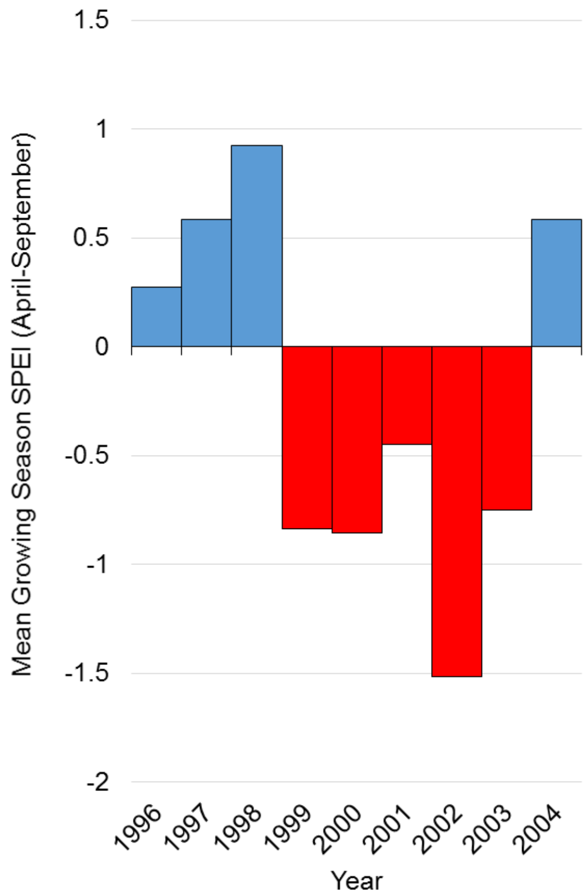

Figure 3.1. Mean growing season (April 1-September 30) Standardized Precipitation Evapotranspiration Index (SPEI) across all sample sites 1996-2004 (Abatzoglou 2013b).

Using a geographic information system (ArcGIS 10.3) I stratified candidate sites within each fire perimeter by north-facing (315° – 45°) vs. south-facing (135° – 225°) aspect and burn severity. Low, medium and high burn severity sites within each fire perimeter were identified using Thematic Burn Severity based on 30-m Landsat satellite imagery from the Monitoring Trends in Burn Severity database (MTBS 2016). Burn severity was verified in the field using measures of overstory mortality, using low (0-20%), moderate (21-70%), and high (71-100%) (Rogan and Franklin 2001, Rother and Veblen 2016). Candidate sites were constrained to within 500-m of a road or trail and slopes < 35° to ensure accessibility. I also excluded areas where post-fire salvage logging and/or replanting had occurred according to the Forest Service Activities Tracking System (FACTS) Database (USFS 2017) and consultation with local forest service staff. I could not stratify sampling by forest type because there were insufficient potential sites due to aspect and burn severity stratification combined with accessibility constraints and widespread post-fire logging and replanting.

In the field, candidate sites were randomly selected for sampling using a random number list. If a chosen site was inaccessible or otherwise did not meet sampling criteria I offset the location by 30-m in cardinal directions to obtain the desired stratification, or selected the next site in the list if this was not possible. Sites were separated by a minimum of 120-m to reduce potential spatial autocorrelation (Kemp et al. 2016).

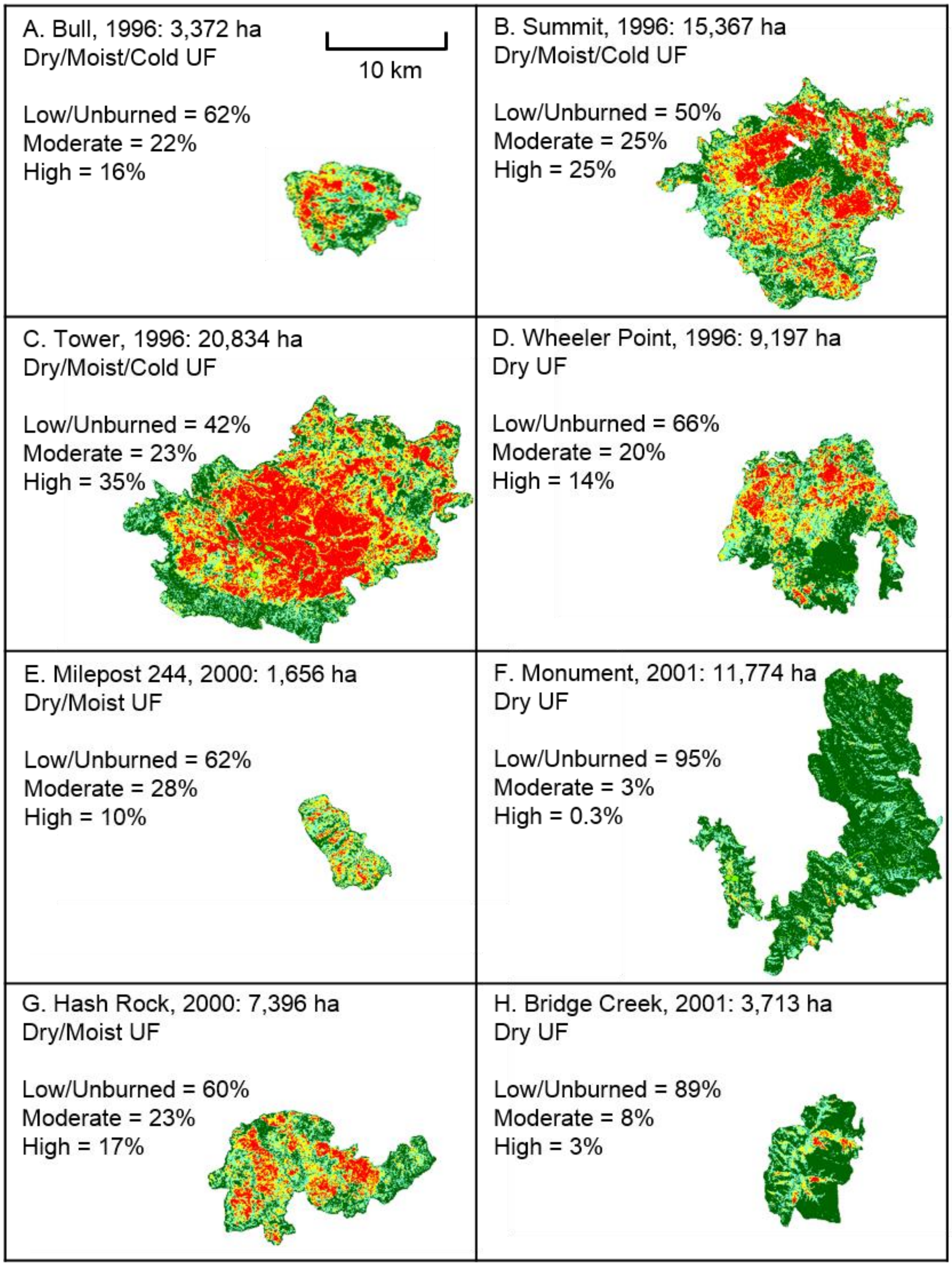

Figure 3.2. Fire characteristics, including year, burn area in hectares, constituent forest types (UF = upland forest) based on plant associations in the burn area, and percentage area burned at low/unburned, moderate, and high severity. Burn severity reflects the Thematic Burn Severity product based on 30-m Landsat satellite imagery from the Monitoring Trends in Burn Severity database (MTBS 2016).

3.3.3 Field measurements

To quantify regeneration at each site I established a belt transect measuring 60-m x 2-m within each using two 30-m tapes parallel to the slope contour. I divided each transect into six sections, 10-m x 2-m, to facilitate data collection. At three points – the center of each transect (30-m) and either end of each transect – I recorded elevation (m), aspect (degrees), slope gradient (degrees), slope position, slope relief, burn severity (low, medium, high) and GPS location to characterize topography and microsite characteristics (Table 1). At each of these three points. I also measured the distance to the nearest living adult tree (potential seed source) using a DISTO laser meter and estimated overstory density using a convex mirror spherical densitometer (Lemmon 1956). I identified species and condition (live, dead but intact, dead broken stem) and measured diameter at breast higher (DBH, cm) of all conifer stem > 12.5 cm DBH tallied using a variable plot with a keyhole gauge to characterize forest structure. I did not have any deciduous trees in my plots, and recorded shrub cover using microplots as described below. I then used these data to calculate basal area and adult trees per hectare for each site. Finally, I took vertical photos of ground cover and canopy structure, as well as photos in all four cardinal directions.

Within each of the six 10-m x 2-m sections I counted all seedlings (> 10 cm high but < 2.5 cm DBH) by species. I also counted all saplings by species (2.5 – 12.5 cm DBH) (USFS FIA Glossary 2016). I could not be certain that seedlings or saplings established post-fire rather than surviving the fire, especially in low severity burn areas, which may have inflated post-fire seedling and sapling counts. However, I found it necessary to count all seedlings and saplings given the significant time since fire. In some high severity burn areas with 100% overstory mortality, saplings that definitely established post-fire (typically lodgepole pine and western larch) exceeded 2.5-cm DBH and were often greater than 2-m high. I wanted to ensure I captured these individuals, particularly to contrast with other burned areas with lower juvenile conifer size and/or abundance.

Within each of the six sections I randomly sampled the five seedlings closest to the center tape and recorded species and measured height (cm) (yielding measurements on up to 30 seedlings per transect). I also recorded the presence of browsing or mechanical damage. I then placed a 0.25-m^2 circular plot in the center of each section and estimated percent cover of shrubs, litter, rocks, bare soil and other ground cover using a modified Braun-Blanquet approach (Rother and Veblen 2016) (Table 1). I measured coarse woody debris (potential seedling nurse sites) by walking along the tape from the center towards each end and recording the distance along the tape, diameter, and length of the first three logs >=7.5-cm diameter (1,000-hour fuels; USFS Wildland Fire Assessment System 2016) that the tape intersected, resulting in a total of six logs measured for each transect. Measurements were recorded on field data forms and entered into a spreadsheet for analysis.

Table 3.1. Predictor variables used to model seedling (> 10 cm high but < 2.5 cm DBH) and sapling (2.5 – 12.5 cm DBH) presence and absence, as well as stocking adequacy.

Category	Variable	Expected relationship	Definition	Importance for conifer regeneration
Fire	Years since fire	+	Years (integer) between fire and sampling year	Seedling recruitment may increase over time (Haire and McGarigal 2010).
	Drought post-fire	+ or −	Mean growing season (Apr-Sept) SPEI of first 3 years post-fire (continuous)	Dry post-fire years may reduce the establishment and growth of juvenile conifers (Stevens-Rumann et al. 2017).
	Distance to seed source	−	Distance (m) from nearest mature, live conifer averaged from points at 0, 30 and 60 m	Conifer seed abundance of wind or gravity-dispersed species declines with distance to seed source (Greene and Johnson 2000).
	Burn severity	+ or –	Percent overstory mortality within ~30 m radius, averaged from points at 0, 30 and 60 m (categorical): (Low: 0-20%; Medium: 21-70%; High: 71-100%)	Low burn severity and correspondingly low canopy mortality may create conditions that are too shaded for regeneration, though may also provide seed and temperature modulation. High canopy mortality may create over-exposed conditions and seed limitation (Fajardo et al. 2006).
Abiotic environment	Elevation	+	Elevation (m) averaged from points at 0, 30 and 60 m	Higher elevations have higher moisture availability and lower temperatures (Dodson and Root 2013).
	Heat load	−	Composite variable derived from slope, aspect and latitude (McCune and Keon 2002)	Topography and latitude determine how heat and dryness may affect seedling establishment and growth (Silen 1960).
	Slope position	+ or –	Slope position (categorical): bottom; lower third; middle; upper third; ridgetop	Slope bottoms may have higher moisture availability, enhancing seedling establishment and growth (Haire and McGarigal 2010).
	Slope relief	+ or –	Topographic relief (categorical): convex; flat; concave	Concave slopes may have higher moisture availability than flat or convex slopes, enhancing seedling establishment and growth (Martín-Alcón and Coll 2016).
Biotic environment	Overstory density	+ or −	Overstory density (%) using convex mirror averaged from points at 0, 30 and 60 m	Overstory density determines the amount of sunlight reaching the forest floor. High densi1ties may create conditions too shaded for seedling growth, while low densities may increase exposure to dryness and heat (Maher and Germino 2006).

	Serotinous response†	+	Binary (0 – absent, 1 – present)	Serotinous lodgepole pine can result in high densities of lodgepole recruitment from the seedbank post-fire.
	Tree basal area	+	Total basal area (m^2 ha^{-1}) of live and dead trees summed from variable radius plots at 0, 30 and 60 m	Basal area is a proxy for site productivity.
Ground cover	Bare Soil	+ or −	Cover class* of bare soil.	Bare soil microsites may be free from competing plants, though potentially exposed to heat, moisture limitation and herbivory.
	Litter	+ or −	Cover class* of litter.	Litter cover reflects crown cover.
	Coarse woody debris (CWD)	+	Cover class* of coarse woody debris.	Coarse woody debris can provide “nurse sites” for germination by increasing soil moisture and decreasing temperature.
	Graminoids	−	Cover class* of graminoids.	Potential competition.
	Forbs	−	Cover class* of forbs.	Potential competition.
	Shrubs	−	Cover class* of woody understory vegetation.	Potential competition and may create overly shaded conditions.
	Rock	−	Cover class* of exposed rock.	Rock is unsuitable for conifer regeneration.

*Cover classed defined as 1: <1%; 2: 1–4.99%; 3: 5–24.99%; 4: 25–49.99%; 5: 50–74.99%; 6: 75–100% (Rother and Veblen 2016).

†Excluded as a predictor variable in models predicting juvenile conifer presence/absence for non-lodgepole sites only.

3.3.4 Data analysis

I used Random Forests (RF) to develop binary classification models using the randomForest (Liaw and Wiener 2002) R package (R Core Development Team). RF is a modeling approach that fits many classification trees to a dataset (Breiman 2001). In a classification tree for a binary response variable such as presence-absence data, the data are recursively split using the predictor variables into nodes that are increasingly homogenous with respect to the response variable as the tree grows (Breiman et al. 1984). RF repeats this process many times using bootstrap samples (e.g. 500) of approximately 63% of the observations ("in-bag" observations) each. Each tree is fully grown using a random sample of predictor variables at each split, and the full tree is used to predict the class of the "out-of-bag" (OOB) observations with the accuracies and error rates averaged across all predictions. In this way, RF essentially produces its own cross-validated accuracy estimates (Cutler et al. 2007). Variable importance can then be assessed, with the most important variables contributing the most to correct classification of the response and the most homogenous terminal nodes on the trees.

I used an RF modeling framework because it has high classification accuracy, makes no distributional assumptions about predictor or response variables, retains collinear predictors, and can accommodate large numbers of predictor variables as is the case with my data (Cutler et al. 2007). I modeled (1) the probability of seedling, sapling, and total juvenile conifer presence or absence in a site for all sites as well as non-serotinous sites only; and (2) the probability that sites on National Forest met local stocking recommendations for specific plant associations. Both classification models were run using the predictor variables in Table 1 (though the "serotinous response" variable was omitted from the non-serotinous models). I tuned the variable sampling parameter in each model by varying the number of predictor variables sampled at each tree split using the "tuneRF" function in R. I selected the parameter value that resulted in the largest decline in the OOB error estimate to run each model.

3.3.5 Stocking adequacy analysis

The topographic heterogeneity of eastern Oregon means site productivity can vary dramatically over short distances. Accordingly, Forest Service stocking recommendations vary from 60 to over 300 trees ha^{-1} for hot-dry to cool-moist plant associations (Powell 1999). Therefore, I obtained local stocking recommendations for specific plant associations to determine stocking adequacy. I used ArcGIS 10.3 (ESRI 2016) to identify the plant association occupied by each of the sites I sampled, specifically using Plant Association Groups sourced from the Field Sampled Vegetation (FSVeg) spatial data (USFS 2017b). I then obtained local stocking recommendations for those plant associations (Powell 1999). Powell's (1999) stocking recommendations are given as a range for specific tree species in specific plant associations. The range is provided as tree densities for the Lower and Upper Limits of the Management Zone, as well as the Full Stocking level. The Lower Limit of the Management Zone (LLMZ) represents approximately 50% of Full Stocking, and densities below the LLMZ are considered understocked. Therefore, as a conservative estimate of minimum stocking densities in mixed species stands, I selected the lowest LLMZ stocking level from all tree species in a given plant association as recommended by (Cochran et al. 1994) (Appendix B, Table B1). Sites were considered understocked if the observed density of juvenile plus live adult conifers was less than the minimum recommended stocking density.

3.4 Results

I surveyed 191 sites in total across eight fires. For data analysis, I eliminated sites where I suspected post-fire salvage logging and/or replanting had occurred based on observed cut stumps or other evidence, even if they were not labelled on available salvage logging/replanting maps. I also eliminated four sites from the Carrol Creek fire, which after two days of surveys was deemed too inaccessible to sample sufficiently. This yielded 184 sites for analysis in which I counted a total of 8,647 individual seedlings and saplings (Figure 3.2). I found wide variation in juvenile conifer densities, ranging from 0 to 87,833 juvenile conifers per hectare (median = 250) (Table 3.2). One-third (33%) of sites contained zero seedlings or saplings, and densities varied across the eight fires surveyed (Table 3.2, Appendix: Figures B1, B2).

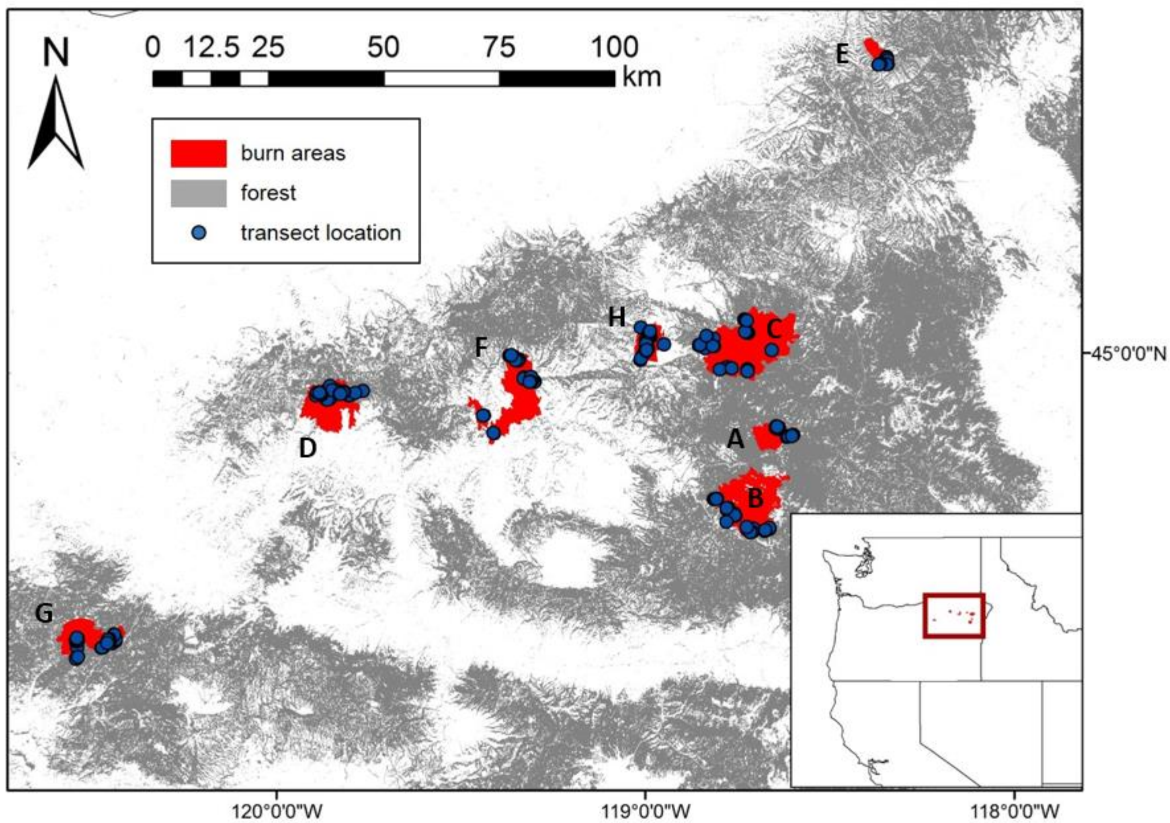

Figure 3.3. Study area with transect (site) locations in the Blue Mountain ecoregion, eastern Oregon, USA. Fires: A) Bull; B) Summit; C) Tower; D) Wheeler Point; E) Milepost 244; F) Monument; G) Hash Rock; H) Bridge Creek.

Table 3.2. Regeneration characteristics from 184 sites across 8 fires.

Years since fire*	Fire (# of sites)	Elevation range (m)	Median seedlings h^{-1}	Median saplings h^{-1}	Median total juvenile trees h^{-1} ǂ	Median species richness ǂ
21	Bull† (15)	1,456 – 1,611	3,500	10,375	14,750	4
21	Summit† (28)	1,209 – 1,503	83	83	125	1
20	Tower† (26)	1,194 – 1,583	542	208	1,208	2
20	Wheeler Point (23)	1,088 – 1,427	83	0	167	1
17	Milepost 244 (15)	863 – 1,361	0	0	0	0
16	Monument (26)	693 – 1,150	208	0	208	1
16	Hash Rock (25)	1,253 – 1,587	417	83	667	1
15	Bridge Creek (24)	871 – 1,144	0	0	0	0

*Survey year – fire year

†Serotinous response by PICO at some sites

ǂSeedlings and saplings

I observed the most abundant regeneration in the Bull and Tower fires, which also had the highest median species richness (Table 3.2). These fires contained most of the serotinous lodgepole sites surveyed (Figure 3.4, Appendix: Figure B2). I felt confident attributing the densities at these sites to a serotinous lodgepole response because they contained nearly pure stands of lodgepole saplings of the same size class in characteristic “doghair densities”, located in areas that experienced high severity fire (70-100% overstory mortality) according to Landsat imagery. In other sites where juvenile conifers were present, transects generally only contained one or two species (Table 3.2). Overall, ponderosa pine and Douglas-fir were the most abundant species, present at 46% and 37% of sites respectively, followed by grand fir/white fir, western juniper, western larch and lodgepole pine (Appendix Table B2). Engelmann spruce and subalpine fir were observed at a few sites at higher elevations. Juvenile conifer seedling species compositions generally aligned with pre-fire forest type based on maps of potential vegetation groups (USFS FSVeg 2017), with the exception of sites where no regeneration was observed, which were dominated by either grasses or shrubs. The fires with the lowest levels of regeneration were the Milepost 244 and Bridge Creek fires, with no regeneration observed in 60% and 70% of sites respectively.

Post-fire understory plant responses ranged from grasses only to shrubs of varying densities, species and heights depending on site productivity. The Milepost 244 fire exhibited a dramatic shrub response comprised of nearly impenetrable mallow ninebark (*Physocarpus malvaceus*) and Scouler’s willow (*Salix scouleriana*), with many plants nearly 3-m tall. Some high-severity burn patches in other fires were completely occupied by snowbrush 1-m to 2-m high (*Ceanothus velutinus*). This species is fire-adapted and the seed coat must be scarified by fire for germination to occur from the seed bank.

I observed multiple different regeneration patterns (Figure 3.5). In low and moderate severity burn areas with living overstory trees providing a seed source, some Douglas-fir, grand fir and western larch seedlings grew at very high densities (Figure 3.5i). Above elevations of 1,200-m in cold upland forest areas I observed serotinous lodgepole pine regeneration, comprising 13% of sites sampled. These “doghair” stands of lodgepole saplings were incredibly dense and nearly inaccessible in some cases (Figure 3.5ii). The maximum density of juvenile conifers excluding serotinous sites was 25,417 juveniles per hectare (median = 167). In larger high-severity burn patches seedlings were often observed in favorable microsites, such as growing from nurse logs (Figure 3.5iii), if observed at all. In several sites observations suggested seedling and saplings were the offspring of large snags which initially survived the fire but died in the intervening years (Figure 3.5iv). These snags illustrate the importance of large, dominant adult trees that can withstand crown fires and serve as residual seed sources.

Figure 3.4. Species composition of juvenile conifers (i.e. both the seedling and sapling size classes) counted within each fire.

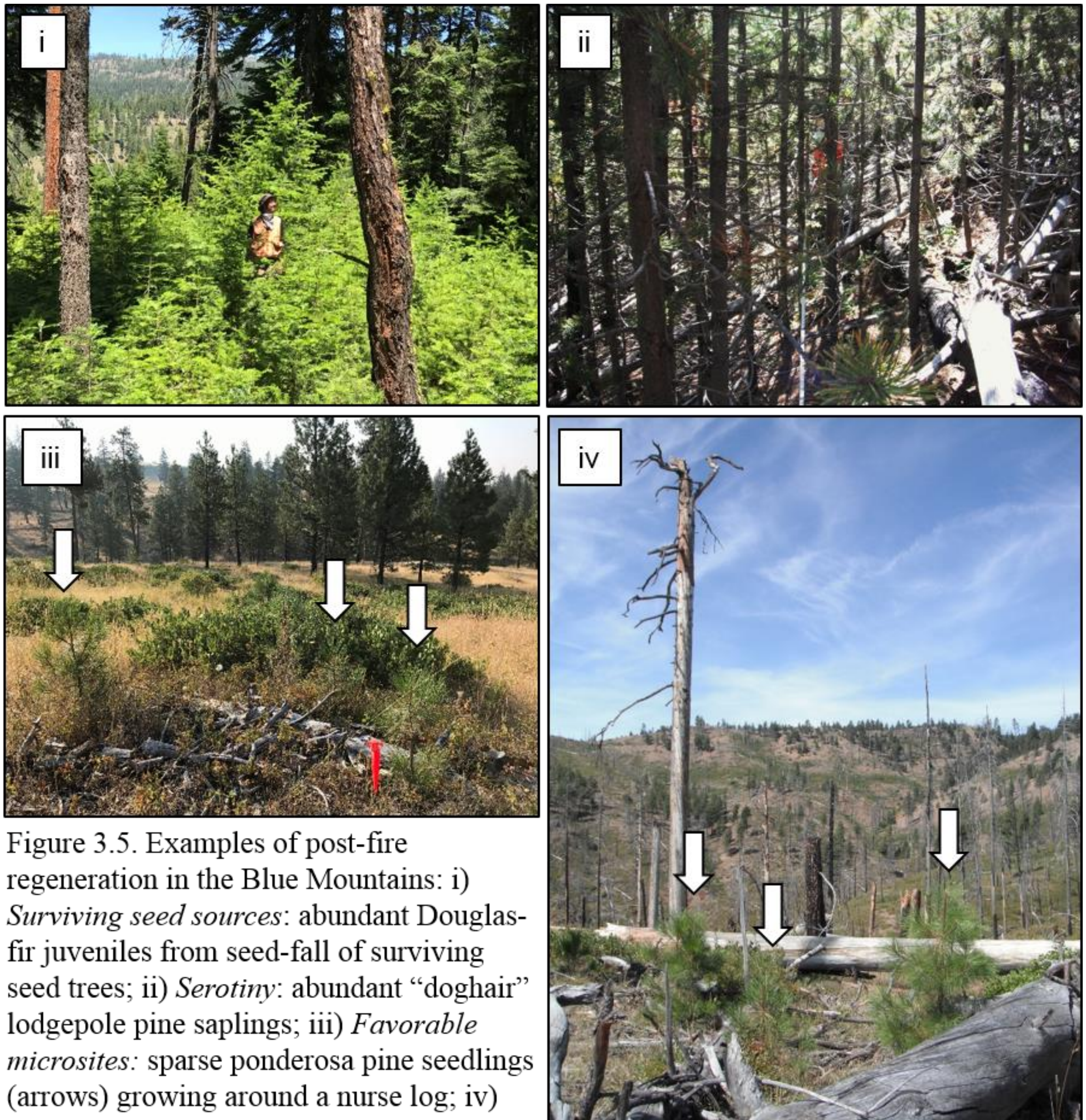

Figure 3.5. Examples of post-fire regeneration in the Blue Mountains: i) *Surviving seed sources*: abundant Douglas-fir juveniles from seed-fall of surviving seed trees; ii) *Serotiny*: abundant "doghair" lodgepole pine saplings; iii) *Favorable microsites:* sparse ponderosa pine seedlings (arrows) growing around a nurse log; iv) *"Ghost" residual seed sources:* in the absence of close living seed sources, these ponderosa pine seedlings are potentially sourced from the snag (background) if the tree initially survived the fire and succumbed later.

3.4.1 Stocking adequacy

I found 111 sites were located on USFS land and within plant associations for which local stocking level recommendations were available. I compared stocking density recommendations for each plant association with the observed densities of juvenile conifers and live adult trees at each site, and found 65% of these sites met minimum stocking recommendations. The few sites surveyed in plant associations dominated by lodgepole pine and subalpine fir in cool upland and moist upland forest respectively met minimum stocking recommendations (Table 3.2). However, over one-third of sites in plant associations dominated by Douglas-fir, grand fir/white fir, and ponderosa pine, in both dry and moist upland forests, did not meet minimum stocking recommendations.

Table 3.3. Percentage of sites adequately stocked with seedlings and saplings on USFS land by aggregated plant associations* (UF = Upland forest), based on local USFS recommendations.

Dominant tree species in plant associations	**Site count***	**Sites above minimum stocking recommendation**
Douglas-fir (Dry/Moist UF)	35	21 (60%)
grand fir/white fir (Dry/Moist UF)	58	38 (66%)
lodgepole pine (Cold UF)	3	3 (100%)
ponderosa pine (Dry UF)	13	8 (62%)
subalpine fir (Moist UF)	2	2 (100%)
Total	111	72 (65%)

***Only sites on USFS land and in plant associations with available local stocking recommendations (Powell 1999) are included here.

3.4.2 Site factors influencing regeneration and stocking adequacy

Seedlings and saplings were more frequently present in transects from the 1996 fires, which were followed by moist conditions in the first 3 years post-fire, than those from 2000/2001, which were followed by drier conditions (Mann-Whitney U, $p < 0.01$). However, this was likely due to differences in forest type and site-level variables and drought was not a consistently important variable predicting presence when accounting for other variables. The results of the Random Forests analyses indicate that the most important variables predicting juvenile conifer presence included heat load, overstory density, distance to seed source, and rock cover. Elevation was the most important predictor of sapling presence for all sites as well as non-serotinous sites only (Table 3.3). Out-of-bag observations were correctly classified 71-82% of the time (correct classifications are the inverse of the error estimates in Table 3.3).

Juvenile conifers were less likely to be found on slopes with high heat loads (Figure 3.5i). Juvenile conifers were more frequently observed where overstory density was higher (i.e. more adult trees at the site) (Figure 3.5ii). Overstory density is strongly correlated with distance to seed source, however both variables were included in the model because overstory density also determines light availability. Juvenile conifer abundance declined further than 200-m from the nearest live seed source (Figure 3.5iii).

Table 3.4. Random Forest model results for binary (0/1) response variables, showing out of bag (OOB) error estimate (% incorrectly classified) and most important predictor variables. The top three predictor variables are shown. See Appendix B for full Variable Importance Plots (Figures B3-B5).

	Model	OOB Error Estimate	Top Variables
All sites (N = 184)	Seedling presence/absence	23.37%	1. Overstory density 2. Heat load 3. Distance to seed source
	Sapling presence/absence	23.91%	1. Elevation 2. Overstory density 3. Heat load
	Regeneration presence/absence (seedling or sapling)	22.28%%	1. Overstory density 2. Heat load 3. Distance to seed source
Non-serotinous sites (N = 159)	Seedling presence/absence	28.93%	1. Heat load 2. Distance to seed source 3. Years since fire
	Sapling presence/absence	27.67%	1. Elevation 2. Heat load 3. Rock cover
	Regeneration presence (seedling or sapling)/absence	28.3%	1. Heat load 2. Distance to seed source 3. Rock cover
USFS sites with stocking recommendations (N = 111)	Minimum stocking present/absent	18.02%	1. Overstory density 2. Rock cover 3. Graminoid cover

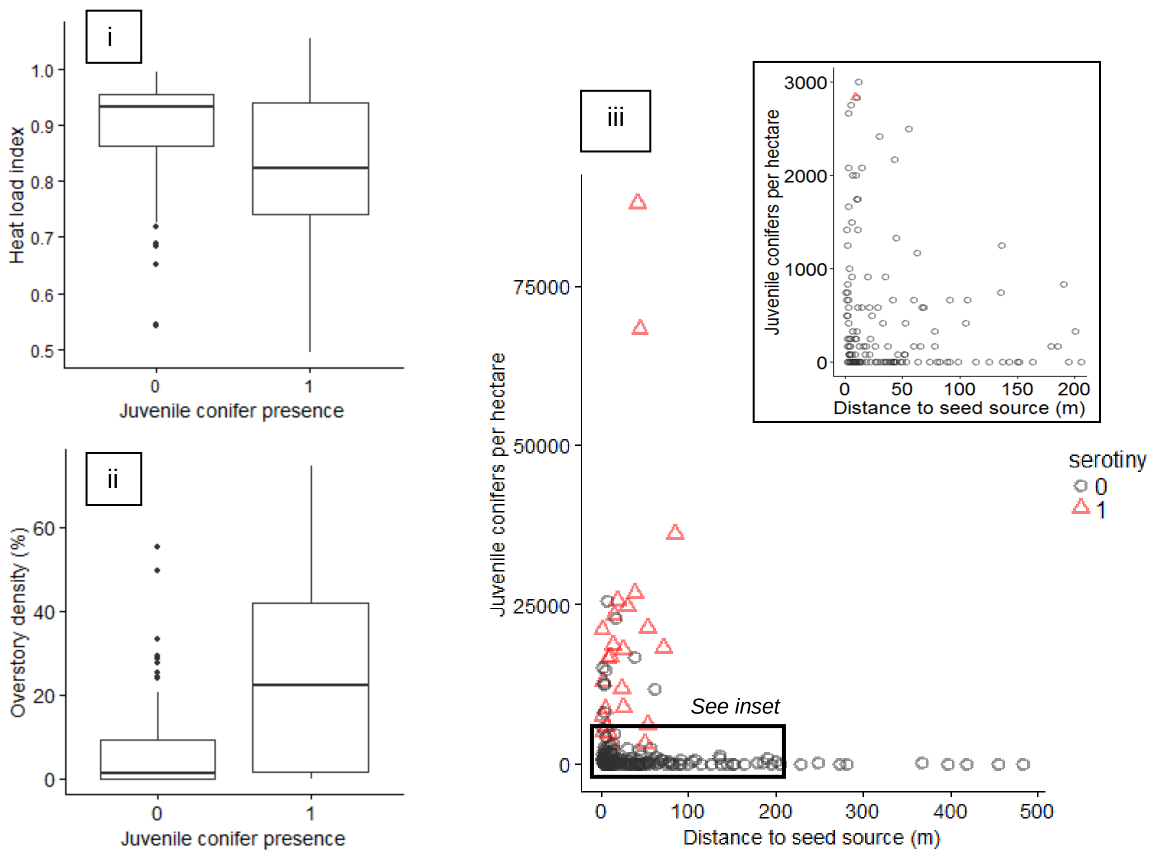

Figure 3.6. Variables influencing post-fire conifer regeneration in the Blue Mountains: (i) the probability of regeneration is lower on warmer slopes; (ii) regeneration is more likely where post-fire crown density is high; (iii) related to post-fire overstory density, the abundance of seedlings and saplings declines dramatically greater than 200-m from a living seed source.

3.5 Discussion

Increasingly frequent large wildfires coupled with climate change may reduce forest resilience in western North America. Already, forest scientists observe regeneration limitation or failure in areas experiencing large high severity burns and/or post-fire drought conditions (Johnstone et al. 2016, Stevens-Rumann et al. 2018). I observed that burn severity patterns and topography are key factors driving conifer regeneration in the Blue Mountains ecoregion. I saw a wide range of regeneration densities across sites and between fires. One-third of sites contained no juvenile conifers 15-21 years post-fire, potentially indicating regeneration limitation or failure, while abundances in other sites were very high and more than sufficient to achieve full stocking levels. The conifer seedling densities I observed align closely with those observed in Montana and Idaho (Kemp et al. 2016). That study, conducted in similar forest types, along with regeneration research in the southern Cascade Range in Oregon, also found differences in regeneration densities of five orders of magnitude (Donato et al. 2009).

3.5.1 Topography and dispersal distance drive regeneration patterns

The dramatic topography of eastern Oregon, which includes steep canyons, mountains, and plateaus with sharp aspect transitions, strongly affects vegetation distributions, fire behavior, and subsequent post-fire forest recovery. I observed regeneration limitation or failure most frequently on slopes with high heat loads (south-facing aspects). High soil surface temperatures can irreversibly damage seedling stem tissue, and damage is most common on flat, south-facing slopes (Isaac 1938, Silen 1960). The mortality rate due to high soil temperatures for Douglas-fir germinants on north-facing aspects can be half or less than that of south-facing aspects (Silen 1960). This may also be why I found overstory density to be an important predictor of regeneration presence. In studies of Engelmann spruce and subalpine fir, shade from tree cover increases soil water content and daily minimum temperatures, enabling higher levels of photosynthesis. In exposed sites, seedlings are vulnerable to nocturnal frost followed by intense sunlight, potentially depressing photosynthesis, growth and survival (Germino and Smith 1999, Maher et al. 2005, Maher and Germino 2006). Other studies have also established a relationship between reduced moisture stress under adult tree canopies and increased regeneration abundance (Bertness and Callaway 1994, Fajardo et al. 2006).

I observed lower probabilities of regeneration greater than ~100-m away from living seed sources. For species with wind-dispersed seeds, seed availability generally declines exponentially with distance from seed source in burn areas (Greene and Johnson 2000). This finding concurs with work in Montana and Idaho where researchers identified a 95-m threshold from residual live seed sources for seedling establishment (Kemp et al. 2016). Other studies in the northern Rockies and Cascade Range also identify distance to seed source as a key determinant of post-fire conifer regeneration (Donato et al. 2016, Harvey et al. 2016).

Saplings were much more frequently observed in sites at higher elevations, indicating more rapid establishment and/or faster growth rates at higher elevations. Dodson and Root (2013) found a strong elevational gradient in conifer regeneration in the eastern side of the Cascade Range in Oregon, observing higher seedling abundances and older seedlings at higher elevations. They observed a strong relationship between elevation and temperature as well as moisture availability, with higher elevations experiencing lower average temperature and higher average precipitation. They concluded moisture-stress is a probable factor limiting conifer regeneration at low elevations in the eastern Cascades, which may also be the case in the adjacent Blue Mountains. In some high elevation sites, I observed serotinous lodgepole pine growing in extremely high densities greater than 2,000 saplings ha^{-1}. Most individuals were more than 2-m tall and many already bore their own cones. Elevation was also the most important predictor of sapling presence for non-lodgepole species, including ponderosa pine, Douglas-fir, western larch and grand fir. These findings suggest high elevation forests in the Blue Mountains, particularly in moist upland and cold upland forest types, have ample regeneration and are currently resilient to wildfire and drought.

3.5.2 Ecological state transitions

While not widespread, I did observe some sites which could be considered ecological state transitions from forest to grassland or shrubland. Regeneration limitation was highest in the Bridge Creek and Milepost 244 fires, where very little regeneration was observed 15 and 17 years post-fire respectively. There were also many sites on the Wheeler Point fire with no regeneration, though juvenile conifers were present at others with nearby legacy seed sources. Most of the sites with limited or no regeneration were located in ponderosa pine and Douglas-fir dominated dry conifer forests at a forest/non-forest ecotone with grassland or shrubland. Forest ecologists observing limited post-fire ponderosa pine regeneration in other regions suggest that post-disturbance replanting may be required for forest restoration (Savage and Mast 2005, Feddema et al. 2013, Collins and Roller 2013). Studies using climate analogs – identifying locations that match the projected climate of other locations in the future – to study shifts in fire regimes and vegetation suggest that dry conifer forests in western North America may be more vulnerable to conversion from forest to non-forest than other forest types (Parks et al. 2017).

The configuration, size, and severity of burns also determines the distribution of legacy seed trees in burned landscapes, dictating subsequent patterns of seedling establishment (Turner et al. 2003). This is likely the reason I observed relatively high regeneration densities on the Monument Complex fire, the other low elevation fire dominated by dry mixed conifer forest. The Monument Complex is a mosaic of unburned, low severity and medium severity burned areas (Appendix: Figure B2(F)) with many residual seed sources. The ponderosa pine and Douglas-fir are widely spaced in many places with grass in the understory, facilitating low severity fire in these fire-adapted woodlands. The area’s topography is also fairly flat, unlike the Bridge Creek or Milepost 244 fires, where steep canyon walls contributed to high-intensity fire behavior.

3.5.3 Study limitations

This study has several limitations that may have affected the results, but may also serve as avenues for future research. Firstly, I could have drawn more robust conclusions about natural regeneration in the Blue Mountains by sampling more sites within each of the region’s various forest types. However, stratifying adequately by forest type was not possible due to accessibility limitations and the prevalence of post-fire salvage logging and replanting. A larger sample size within each forest type may have also more adequately tested the effects of post-fire climatic differences, given that more of the fires from 1996 occurred in cool-moist forest than those from 2000/2001, where lower regeneration would be expected overall. However, there were no other sufficiently large fires in the region from these two periods to sample. Additionally, my analyses do not rigorously compare pre-fire tree species composition to post-fire juvenile conifer composition, and therefore I cannot make detailed statements about species changes. While I have data on species composition of living trees in sites, in many sites adult trees killed by fire were so charred that the species could not be identified. I also do not have local data on mast years. Masting events could have contributed to regeneration pulses in certain areas of masting species such as ponderosa pine and Douglas-fir. Finally, the clumped nature of some conifer regeneration means that my randomly located belt transect sampling technique may have underestimated the abundance of juvenile conifers on the landscape (Fajardo et al. 2006, Dodson and Root 2013). While I did not quantify the spatial distribution of seedlings, a similar study in the southern Cascades determined seedling distributions were random overall rather than clumped (Donato et al. 2009).

While there is growing consensus that regeneration limitation or failure is occurring in some areas, disagreements about what qualifies as regeneration failure remain, potentially affecting the conclusions of this study and others. Haire and McGarigal (2010) aged second-growth ponderosa pine from fires in the 1970s in New Mexico and Arizona, and determined that few seedlings naturally established less than 15 years post-fire. They concluded that low regeneration rates observed 10-20 years post-fire may not be very different from historic rates and therefore that these fires should not be viewed as catastrophic and replanting may not be necessary. It may be that the burned areas I surveyed will “fill in” adequately with natural regeneration in the coming decades. More research is needed to understand the threshold between slow but continued recovery and ecological state transitions.

3.5.4 Management implications

This study provides local forest managers with baseline information on which they can base long-term monitoring for ongoing adaptive management. While I only assessed stocking sufficiency for certain sites on US Forest Service land where stocking recommendations were available, the results can likely be applied to the same plant associations on non-Forest Service land. My results suggest one-third to 40% of post-fire naturally regenerating sites on private, state and BLM lands may be understocked. This suggests more replanting may be necessary following wildfire if there are large areas in which managers are relying on natural regeneration, particularly on hotter sites in large high severity burn patches. However, this recommendation comes with a caution. Stocking all potentially forested areas may be undesirable. As Hessburg et al. (2015) and others have emphasized, a fire- and climate-adapted forest landscape is a heterogeneous forest landscape consisting of patches of forest and non-forest. Historically, this meant few trees on south-facing slopes and at lower elevations in much of the inland Northwest and generally larger areas of non-forest (Hessburg et al. 2005). In addition to making forest landscapes more resilient, the open habitats found in heterogeneous post-fire landscapes make important contributions to biodiversity (Lindenmayer and Franklin 2002).

South-facing slopes are likely to become increasingly inhospitable for tree growth as the climate continues to change. Therefore, some recent large wildfires may present an opportunity to restore the heterogeneous forest structure perpetuated by pre-European fire regimes (i.e. fires at higher frequency with smaller high severity burn areas and more heterogeneous burn patterns). Managers must strike a balance between embracing fewer trees on the land, while staying conscious of increasingly likely ecosystem transitions. Like the overabundance of trees contributing to destructive recent wildfires, forest loss in the western US also raises a host of concerns regarding carbon sequestration, wildlife habitat, and ecosystem services such as water quality and quantity management. Parks et al. (2017) estimate that current climate projections could result in the loss of 12% of current forest area by 2100 as marginal conifer forest is replaced by shrub or grasslands.

Many of the fires surveyed here had large high severity burn patches (Figure 3.1), and recent fires suggest this trend will continue in the Blue Mountains. Widespread overstory removal due to logging in the late 19^{th} and throughout the 20^{th} century, combined with fire suppression and overgrazing, have contributed to dense, young forests in many parts of the region. In the last 20 years large crown fires have been frequent, as illustrated by the 2015 Canyon Creek Complex fire. The fire was over 40,000 ha in size, and over 10,000 ha (26,000 acres; 26% of total burn area) burned at high severity. The loss of vegetation has raised significant flood concerns in local communities (Blue Mountain Eagle 2017). Local managers may not be able to rely on natural regeneration for forest recovery in these recent mega fires, particularly at lower elevations near communities.

3.6 Conclusions

This study emphasizes the importance of topographic controls on post-fire conifer regeneration. As the climate warms though the 21^{st} century, warmer aspects at low elevations may be most vulnerable to ecological state transitions following disturbance. This study suggests moist and cold mixed-conifer forest types are currently resilient to wildfire in the Blue Mountains, though findings from Chapter 2 suggest this resilience may decline over the 21^{st}

century. Dry forest types may experience post-fire regeneration limitation at the forest – non-forest ecotone. Region-specific studies like this are important for elucidating the local factors affecting post-fire forest recovery. Post-fire vegetation monitoring will continue to be an important tool for adaptive forest management as climate and wildfire regimes continue to change.

CHAPTER IV

CLIMATE CHANGE BELIEFS AND FOREST MANAGEMENT IN EASTERN OREGON: IMPLICATIONS FOR INDIVIDUAL ADAPTIVE CAPACITY

4.1 Abstract

Climate change adaptation in most sectors requires understanding the decisions of private actors. In the forest sector, the management decisions of private landowners affect forest structure and composition, and may impact the resilience of forested regions. In this case study I assessed barriers to both intentional and incidental climate-adaptive forest management among non-industrial private forest owners in eastern Oregon, USA. In this context, incidental adaptations result from synergies between climate-adaptive forest management and actions motivated by goals such as wildfire mitigation, which landowners may prioritize regardless of concerns about climate change. Through semi-structured interviews I used qualitative analyses to identify barriers to adaptation, including subjective (cognitive and experiential) and structural barriers (social, political, and economic) by comparing individual cases. Overall, I found that intentional climate change adaptation had low salience among participants, though a large majority of forest owners were active managers motivated by other goals, contributing to widespread incidental adaptation. I found that non-industrial private forest owners who engaged in or considered intentional climate adaptation actions generally believed that anthropogenic climate change is occurring. Many respondents perceived local environmental change – notably reduced snowpack – but this was not associated with adaptive actions or intentions. The few

participants who considered or implemented intentional climate adaptation actions generally had written forest management plans containing both forest inventories and specific management goals. Improving access to resources for forest management planning may enhance fire- and climate-smart forest management by facilitating scenario visioning and formalizing intentions. While climate change beliefs were subjective barriers to intentional climate adaptation, many of the same structural barriers limited intentional and incidental adaptation. Place-based education, reliable funding mechanisms, and cooperative approaches among landowners may enhance adaptive capacity and promote the resilience of these non-industrial private forestlands.

4.2 Introduction

As climate change adaptation theory develops there is growing interest in understanding the conditions that provide opportunities for and barriers to adaptation among institutions and individuals (Moser and Ekstrom 2010, Biesbroek et al. 2013, Klein et al. 2014, Eisenack et al. 2014). Adaptation is the process of adjustment to actual or expected climate and its effects that may or may not moderate harm or exploit beneficial opportunities (modified from IPCC 2014). Adaptation actions can take various forms. They include technological projects or social reforms that reduce the exposure and sensitivity of ecosystems and communities to climate variation and increase their adaptive capacity (Leichenko and O'Brien 2006). An individual's adaptive capacity describes their ability to respond successfully to climate variability and change based on adjustments to behavior, resources and technologies (modified from Adger et al. 2007). Barriers to adaptation are impediments to specific adaptation actions that can be reduced or overcome (modified from Eisenack et al. 2014). Evidence indicates that individual landowners are beginning to consider adaptation actions in forestry in North America and Europe, with self-reported rates of implementation varying from very low to moderate (Keskitalo et al. 2011, Blennow 2012, van Gameren and Zaccai 2015, Bissonnette et al. 2017, Vulturius et al. 2018). Forest management decisions have decadal and centennial-scale repercussions for forest landscapes, enhancing the importance of understanding opportunities for and barriers to climate-adaptive forest management (Lawrence and Gillett 2011, Schoene and Bernier 2012, van Gameren and Zaccai 2015).

Factors promoting or constraining individual adaptation can generally be characterized as subjective barriers resulting from cognitive and experiential processes, or structural barriers arising from broader economic, social, or political conditions (Grothmann and Patt 2005, Smit and Wandel 2006, Vulturius et al. 2018). Subjective barriers include beliefs about the existence of anthropogenic climate change and its effects (Blennow 2012), which may be conditioned by cultural orientation (Kahan and Peters 2011). Climate change beliefs in turn influence individual local or global concern about climate change and relative perception of risk (Slovic et al. 2007, van der Linden 2015). Relative risk perception is the perceived probability of being exposed to climate change impacts and an appraisal of how harmful those impacts will be to things the individual values (Grothmann and Patt 2005, van der Linden 2015). Immediacy of harm is also an issue; most individuals respond to concerns that are immediately and personally relevant (Paton et al. 2001, Moser and Dilling 2004, Adger et al. 2009). An individual's perception of their own ability to adapt, which may or may not match their objective capacity, may also form a barrier to adaptation (Grothmann and Patt 2005, Tompkins and Eakin 2012). Finally, experiential factors, including experiencing or perceiving extreme weather events, may influence risk perceptions and levels of concern, contributing to action or inaction (Amundsen et al. 2010, Blennow 2012, Akerlof et al. 2013).

Structural barriers to climate change adaptation include the political, social, environmental, and economic constraints on individual adaptation decision making and implementation (Smit and Wandel 2006, Moser and Ekstrom 2010, Eisenack and Stecker 2012, Biesbroek et al. 2013, Klein et al. 2014). Individuals may lack the financial, social or political capital to perform climate-adaptive management actions. Individuals may also find themselves in broader institutional or governance contexts with inadequate leadership, communication or information (Moser and Ekstrom 2010, Biesbroek et al. 2013). The importance of subjective versus structural barriers to adaptation depends on context, and a key priority in adaptation research is understanding which barriers arise in certain contexts in order to inform interventions (Wise et al. 2014).

In the developing literature on climate change adaptation by individual private forest owners, studies diverge on the relative importance of structural versus subjective barriers to adaptation. Forest management also highlights the synergies and differences between current adaptive strategies to address existing risks, including insects, disease, wildfire and storms, and adaptation options to enhance resilience to climate change. This study aims to explicitly consider individual adaptive capacity in both contexts. In the next section I review the contrasting evidence on important barriers to adaptation among private forest owners, then differentiate between existing and climate change-adaptive forest management actions in western North America. I then describe my case study, research methods, and findings on private non-industrial forest owners in eastern Oregon. The final section situates my findings within the broader literature on individual climate change adaptation and provides recommendations for communication and policy in western North America.

4.2.1 Climate change adaptation by individual forest owners

Climate change is altering forest ecosystems globally. Rising temperatures and shifting moisture regimes are gradually shifting growing conditions for tree species, while climate-mediated changes in the frequency, size and intensity of disturbances will continue to alter forest structure and function in decades to come (Turner 2010, Vose et al. 2012, Enright et al. 2015). Climate-mediated forest disturbances such as wildfires, insects and disease outbreaks ignore property boundaries and spread across both public and private lands. In the US and Europe over fifty percent of forested lands are privately owned (both non-industrial and industrial), and therefore private management responses to climate change may impact the socioecological resilience of forested regions (UNECE FAO 2010, Tompkins and Eakin 2012, Ruseva and Fischer 2013, Butler et al. 2016).

Non-industrial private forest owners (hereafter forest owner(s)), also known as family forest owners, control 36% of forested lands in the US and typically own smaller tracts of forest compared to large commercial (or industrial) timber estates (Butler et al. 2016). In the western US, these non-industrial private lands are often at low elevations near towns and rural communities (Latta et al. 2010), and are thus the “front lines” for buffering communities from climate change-related natural hazards, including increasingly frequent large wildfires (Westerling 2016, Abatzoglou and Williams 2016). Non-industrial private lands often border federal or state lands, creating a mixed-ownership landscape in which their management practices affect the continuity of fuels, and therefore wildfire, between public lands and communities (Ager et al. 2012, Fischer and Charnley 2012).

Many studies assess the motivations behind private landowner wildfire mitigation actions such as fuels management (reviewed by McCaffrey et al. 2012). The rich literature on barriers to fuels management and prescribed burning indicates that management preferences do not appear to be influenced by demographic characteristics; decisions to implement wildfire mitigation actions are influenced by social context, trade-offs with other amenity values, perceived efficacy of activities, and personal capacity to implement management actions (McCaffrey et al. 2012). People who perceive greater social capital in their community are also more likely to take action on their properties to reduce wildfire risk (Agrawal and Monroe 2006). Additionally, in a study conducted in central Oregon researchers found that landowners’ perceptions of wildfire risk and propensity to conduct fuel treatments correlated with hazardous fuel conditions near their land, whether they have a home at risk, prior experience with wildfire, financial capacity to conduct treatments, and membership in land stewardship organizations (Fischer et al. 2014). These drivers of and barriers to fire-smart forest management echo those identified in the climate adaptation literature (Spies et al. 2010). Indeed, most studies identifying barriers to climate adaptation identify barriers that are not climate change-specific, but rather represent existing challenges in natural resources management (Biesbroek et al. 2013)

Climate change adaptation within the context of forest management can include intentional and anticipatory climate change-specific responses such as planting tree species that will be better adapted to a future climate (Yousefpour et al. 2017), or managing forest density and composition outside of the historic range of variation (Keenan 2015, Nagel et al. 2017). Responses can also be reactive "wait-and-see" approaches to cope with nascent threats, such as cutting trees that appear water-stressed (Beck 1992, Yousefpour et al. 2017). Some responses can be both anticipatory and reactive, such as thinning stands to improve water capture, storage and flow (Grant et al. 2013). Thinning may mitigate both current and future drought stress, wildfire spread, and disease and insect outbreaks, all of which may increase under climate change in certain regions (Chmura et al. 2011, van Gameren and Zaccai 2015). Thinning may therefore represent a "no-regrets" or "win-win" adaptation option (Carter 1996), addressing both current and future risks. Similar actions include creating defensible spacing, underbrush clearing, and using prescribed burns to return historic fire regimes to ecosystems (Clark et al. 2016).

Adaptation may therefore be incidental if landowners carry out such actions for reasons not primarily related to climate change. I use the term "incidental" as opposed to "accidental", as used elsewhere (van Gameren and Zaccai 2015), because I feel that "accidental adaptation" suggests a complete lack of awareness of climate change. In reality, the importance of climate change in motivating forest management actions falls along a continuum from high to low importance depending on the individual and interacts with perceptions of risk associated with existing threats to forests, as demonstrated in other studies of private forest owners (Bissonnette et al. 2017, Vulturius et al. 2018).

A growing number of studies investigate forest owner intentions and actions on climate change adaptation (Blennow and Persson 2009, Blennow 2012, Blennow et al. 2012, Grotta et al. 2013, Lawrence and Marzano 2014, van Gameren and Zaccai 2015, Sousa-Silva et al. 2016, André et al. 2017, Bissonnette et al. 2017, Vulturius et al. 2018). In Sweden, approximately 20% of non-industrial private forest owners reported adapting their forest management to climate change in some way (Blennow and Persson 2009, Blennow 2012) while 40% reported an intention to adapt (Vulturius et al. 2018). Nearly half of forest owners in Germany and Portugal reported implementing adaptation actions (Blennow et al. 2012). Additional case studies suggest low engagement with adaptation in Wales and moderate levels of engagement in Belgium (Lawrence and Marzano 2014, van Gameren and Zaccai 2015). Researchers classified half of Belgian interview respondents as climate change-motivated adaptors and one-third as incidental adaptors (van Gameren and Zaccai 2015). Across all studies, the most commonly reported adaptation actions were increasing tree species diversity and diversifying age structures, in addition to some intentional selection of future climate-adapted species (Blennow 2012, van Gameren and Zaccai 2015, Sousa-Silva et al. 2016, Bissonnette et al. 2017).

The importance of structural versus subjective barriers to climate change adaptation among private forest owners appears to depend on context. Studies from Sweden indicate cognitive factors, specifically strength of belief in anthropogenic climate change, overwhelmingly predict adaptation intention or action (Blennow and Persson 2009, Vulturius et al. 2018). There is also evidence that those with strong climate change beliefs who are not adapting perceive they have low adaptive capacity (Blennow and Persson 2009, van Gameren and Zaccai 2015). In contrast, in Belgium belief in climate change is a poor predictor of adaptation, likely because levels of belief among private forest owners are universally high. Instead, the structural barrier of poor access to technical information on adaptation is the most important factor constraining adaptation (van Gameren and Zaccai 2015). In Quebec, eastern Canada, despite the majority reportedly perceiving anthropogenic climate change, three quarters of landowners perceive limited or non-existent impacts on forests in the short and medium term and feel no need to adapt (Bissonnette et al. 2017). Finally, a series of forest owner focus groups across the US Pacific Northwest and Alaska in 2009-10 found that very few reported changing or adapting forest management practices in response to climate change (Grotta et al. 2013). To my knowledge there are no other studies of climate change adaptation among individual private forest owners in western North America.

4.2.2 Adaptation in conifer forests of the western USA

My case study investigates climate change adaptation among non-industrial private forest owners in Oregon's Blue Mountain ecoregion in the Inland Pacific Northwest, USA. Warming-induced declines in snowpack are expected to increase the frequency and intensity of drought stress, reduce tree growth and survival, increase disturbance by wildfire, insects and disease, and change forest composition and structure (Chmura et al. 2011, Spies et al. 2014, Halofsky and Peterson 2016). The annual percent area burned is projected to increase by 36% assuming fire suppression is maintained under a high emissions scenario (RCP 8.5) (Sheehan et al. 2015, Dalton et al. 2017).

Table 4.1 shows common recommendations for adaptation in dry mixed conifer forests of western North America to respond to these changes (Chmura et al. 2010, Keenan 2015, Halofsky and Peterson 2016). I considered whether these recommended actions could result from intentional or incidental pathways to adaptation, and concluded that several recommended adaptations could result from both. There are additional recommendations for regional-level climate-adaptive forest management (e.g. Hessburg et al. 2015), but here I focus on feasible actions for individual landowners.

Table 4.1. Overview of property-level climate change adaptation recommendations from the literature for dry mixed conifer forests in western North America. Int: Outcome of intentional climate change adaptation. Inc: Outcome of incidental adaptation.

Property-level forest management recommendations			
Structure		Composition	
Density management	Thinning (Int/Inc)	Assisted migration	Traditional or molecular breeding to alter within-species genetic composition (Int)
	Thinning to wider-than-historic spacing (Int/Inc)		Selecting provenances/species for retention or replanting that are "future-adapted" for a given site type (Int)
	Replanting following logging or natural disturbances at lower densities (Int)		
Fuels reduction	Manual/mechanical fuels removal (Int/Inc)	Diversification	Planting or maintaining multiple species as a "bet-hedging" strategy (Int/Inc)
	Prescribed fire (Int/Inc)		

4.2.3 Study goals

A 2014 telephone survey of the general public in eastern Oregon found that although 84% of respondents say they believe climate change is happening, they are roughly split on whether current changes have human (43%) or natural (41%) causes (Boag et al. 2015). These causal beliefs are highly politicized, and generally align with liberal and conservative political affiliations respectively (Hamilton et al. 2016, 2018, Dunlap et al. 2016). Doubt regarding the anthropogenic causes of climate change is widespread in other non-coastal areas of the western US (Howe et al. 2015). At the same time, large wildfires have become more frequent in recent years, partly due to climate change, but also because of high fuels loads caused by 20th century and ongoing fire suppression (Hamilton et al. 2016, Abatzoglou and Williams 2016). Therefore, forest owners in the western US exist in a social and environmental context which may complicate climate change risk perceptions and motivations for adaptation. Focusing on wildfire risk mitigation through fuels reduction may simultaneously achieve climate adaptation benefits.

Using an actor-centered perspective recommended by Eisenack et al. (2014), this study aims to advance understanding of opportunities for and barriers to adaptation among forest owners. I orient this investigation around the three general phases of adaptation (Moser and Ekstrom 2010). The understanding phase involves problem detection and information gathering; the planning phase involves developing and selecting adaptation options; and the management phase involves implementation, monitoring and evaluation. This orientation is useful because it tracks forest management decisions from idea formation through evaluating options to prioritizing and taking actions, while recognizing that decision makers do not always progress sequentially from one stage to the next (Moser and Ekstrom 2010).

First, I identify the adaptive actions non-industrial private forest owners in eastern Oregon have taken or intend to take. I also consider whether those actions are anticipatory or reactive to climate change and/or intentional or incidental. I argue this distinction is important when evaluating adaptation, because it distinguishes between incremental adaptation designed to help systems resist or be resilient to ongoing threats, versus transformational adaptations in the form of new practices designed to prepare systems for future threats that are outside the historic range of variability (Klein et al. 2014, Yousefpour et al. 2017). Second, I identify existing structural and subjective barriers to those adaptive actions, and compare my findings to those from other studies of private forest owners to understand which are most important for specific intentional or incidental adaptation actions. I hypothesized that individuals who do not believe anthropogenic climate change is occurring would not engage in intentional climate change adaptation, representing a subjective barrier. But, I hypothesized that they may demonstrate high levels of incidental adaptation due to high wildfire risk in the region. I also hypothesized that most of the structural barriers to adaptation would be non-climate specific and similar to those identified in other regions with low economic and institutional capacity for active forest management. I supplement my qualitative analysis with excerpts from participant interviews to provide a richer picture of their perspectives for researchers, non-profits, agencies and peers working to support climate-smart forest management.

4.3 Methods

4.3.1 Study area

I conducted interviews with non-industrial private forest owners in eastern Oregon, USA in the Blue Mountains ecoregion, which extends into parts of Idaho and Washington states. Oregon's largest ecoregion is comprised of rugged mountains and steep valleys and plateaus, with elevations ranging from 500 to over 3,000 m. Temperatures vary widely between seasons, with winter lows below -15°C and summer highs above 35°C (NOAA 2017). Average annual precipitation was 40 cm over 1981–2010, with most precipitation falling in winter and spring as snow and rain and little precipitation in summer and fall, the region's fire season (NOAA 2017). Dominant forest types in warm/dry sites and at low elevations are ponderosa pine (*Pinus ponderosa*) and warm mixed-conifer forests, while cool mixed-conifer and lodgepole pine forests (*Pinus contorta*) exist on cool/wet sites and at high elevations (Emmingham et al. 2005). Approximately 71% of eastern Oregon forests are federally owned, while 27% are privately owned and 2% are managed by nonfederal agencies (Campbell et al. 2003)

I performed all interviews in four counties (Crook, Wheeler, Grant, and Wallowa — Figure 4.1), selected for their diverse social and ecological contexts. Furthest west, Crook, Wheeler and Grant counties lie in the rain shadow of the Cascade Mountains. These counties have cool mixed-conifer forest at high elevations, large swaths of dry mixed conifer and ponderosa pine forest at mid elevations, and western juniper (*Juniperus occidentalis*) woodlands and shrub steppe at lower elevations. Grant is the only county that still has a working lumber mill, while Crook and Wheeler counties are closest to large population centers like Bend. Wallowa County in Oregon's northeast corner is cooler and wetter with larger areas of cool mixed-conifer forest at lower elevations. Wallowa also has a larger proportion of seasonal property owners and a larger tourism industry. Across all four counties, most private forestland is dominated by ponderosa pine and Douglas-fir (*Ptsuedotsuga menziesii*), in addition to grand fir (*Abies grandis*) and lodgepole pine in mixed conifer sites.

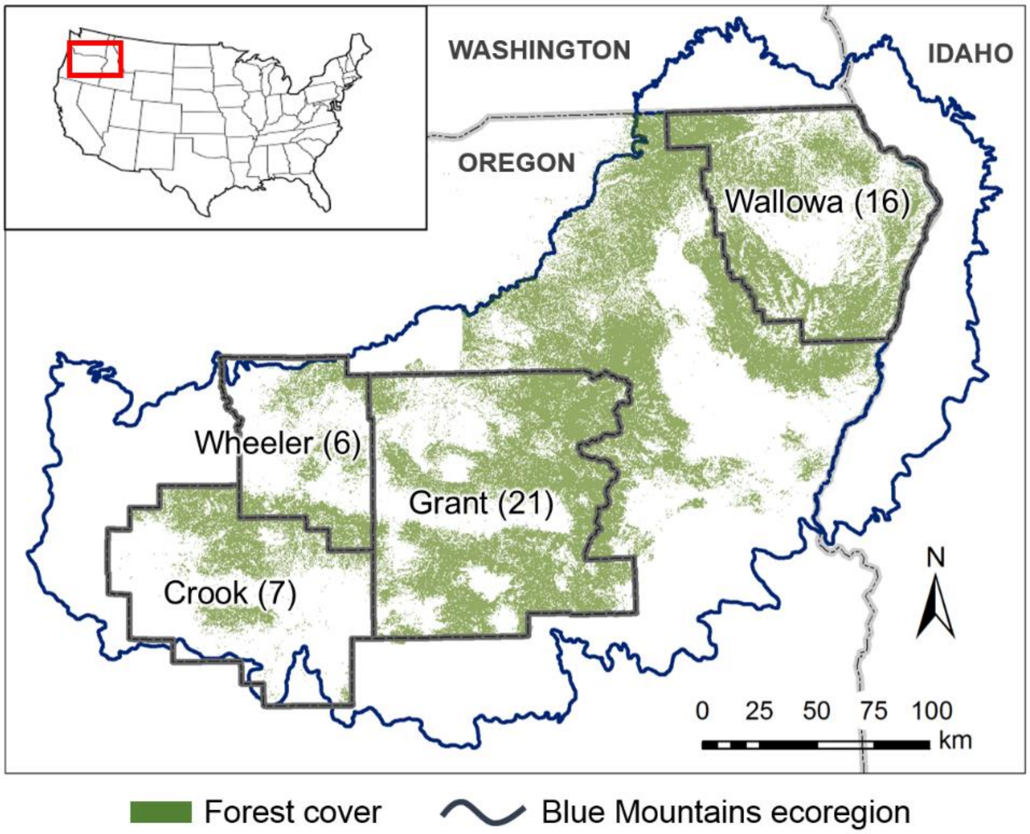

Figure 4.1. Study area showing number of interviews (in parentheses) conducted in each county.

In eastern Oregon climate change impacts manifest as more frequent large wildfires, earlier springs, longer fire seasons, rising summer temperatures and declining snowpack (Halofsky and Peterson 2016, Hamilton et al. 2016, Abatzoglou and Williams 2016). Fire suppression and overstory logging over the 20th century have also contributed to more frequent large wildfires, as in much of the US West. Manufacturing, forestry, mining and agriculture founded the area's economy, all of which continue to have a strong presence in the culture of local communities despite seeing significant declines in the last 30 years. Dramatic declines in logging in the 1990s due to policy changes on federal lands and other factors have shrunk the region's forest products industry infrastructure, including its workforce and log hauling capacity (Christoffersen 2005).

4.3.2 Sampling frame and interview protocol

I conducted an extensive literature review focusing on regional climate change impacts, wildfire history, as well as rural and environmental sociology to inform the development of a semi-structured interview guide (Appendix C), and conducted a pilot interview to refine my questions (Patton 2002). I identified landowners in each of the four counties owning greater than 4 ha (10 acres) forested land using publicly available tax lot data. I mailed them letters (N = 417) inviting them to participate in the study (response rate = 7%). The lead author then identified further participants through snowball sampling (Patton 2002), asking those who responded to my mailers to suggest other forest owners who may be interested in participating. Therefore, my participants represent a non-random self-selected group who are likely more interested and engaged in forest management than typical non-industrial private forest owners in this region.

I conducted all interviews to maintain topical consistency across all subjects and allow participants to respond in as much detail as they wished. Interviews covered six main topic areas: 1) Management goals; 2) Forest management planning; 3) Management activities, both ongoing and planned; 4) Perceptions of local wildfire risk, drought, changes in snow and precipitation, and forest condition; 5) Beliefs and attitudes regarding climate change; and 6) Engagement and resource needs. Participants were also asked to fill out a questionnaire on their demographic information. I intentionally asked about beliefs and attitudes regarding climate change near the end of the interviews to facilitate an open discussion of management goals, plans, actions, concerns, and perceptions of environmental change, and to maintain trust between the interviewer and participants.

A total of 50 landowners were interviewed between June and August 2015 (Wallowa county: 16; Grant county: 21; Wheeler county: 6; Crook county: 7). Interviews lasted from 45 minutes to two hours in length. Each respondent could decline to answer any question and end the interview at any point, though no one ended the interview prematurely. The interviewer volunteered to visit subjects on their properties, and in most cases the interviewer toured subjects' forest property with them during the interview. Interviews were audio-recorded with respondents' permission and also documented using extensive written notes. The audiotapes and interview notes were then transcribed for analysis.

Of the 50 respondents interviewed 72% lived on their property year-round, and most were well established in their communities. Many had lived in eastern Oregon for over 30 years (median = 37), and owned their properties for more than 20 years (median = 25). Property sizes of participants ranged from 6 to 5,000 hectares (median = 57 ha) (Table 4.2). Fifty-eight percent of respondents grazed cattle on some or all of their land (either their own cattle or leasing pasture to others), both in forested areas and pastures, and 30% of respondents also practiced other forms of agriculture (e.g. hay, crops such as canola and wheat, fruit orchards).

Table 4.2. Demographic characteristics of participants (N=50).

Median age (years)		66
Percent Male		80%
Median property area (ha)		57
Property acquisition	Inherited	26%
	Purchased	74%
Plan to leave in 5 years	No	96%
	Yes	4%
Education	Less than High School	2%
	High School	14%
	Some College	26%
	College	44%
	Graduate School	14%
Political orientation	Democrat	12%
	Independent (lean Democrat)	28%
	Independent	18%
	Independent (lean Republican)	12%
	Republican	30%

4.3.3 Data analysis

I used a qualitative research approach because of its suitability for exploring the unique perceptions and activities of individual forest owners, particularly in the context of the emerging phenomenon of climate change adaptation (Bliss and Martin 1989, Creswell 2013, van Gameren and Zaccai 2015). I analyzed interview responses using inductive grounded theory (Glaser and Strauss 1967, Strauss and Corbin 1990), iteratively coding for emergent patterns and themes using NVivo 10.1 (QSR International). In contrast with a hypothetic-deductive approach, in grounded theory repeated themes and concepts are categorized, then these categories are compared and contrasted to form the basis for new theory (Strauss and Corbin 1994).

Based on the methods of a similar study by Nicholas and Durham (2012), answers to each question were grouped across all participants and initially coded using a pool of themes generated from 5 interviews. Interviews were coded for emergent themes and subthemes related to implemented or envisaged adaptation actions, overall forest management planning and actions, perceptions of local environmental change, beliefs regarding climate change, and resource limitations. Following the initial coding round, themes and subthemes were added, combined or eliminated as needed in coding the rest of the interviews (Miles and Huberman 1994). After coding was complete, all transcripts were checked against the final code list (Appendix C) to ensure all essential concepts were captured. I then compared individual cases by constructing qualitative data matrices in NVIVO to explore patterns and connections between themes relating to cognitive and experiential factors, objective adaptive capacity, and forest management actions, including incidental and intentional adaptation actions.

4.4 Results

4.4.1 Forest management activities

I asked respondents to describe their overall management goals for their land, and most landowners had multiple goals. The three most common goals reported by approximately half of the participants (either individually or in combination) were enhancing timber growth and yield, providing wildlife habitat, and developing and maintaining alternative amenity values. These alternative amenity values, which I coded as a single theme, included scenic value, solitude, aesthetics and recreational opportunities. Approximately one quarter of participants had specific overall goals relating to habitat or forest restoration (including reducing wildfire risk), and procuring firewood. A small minority owned their land as a financial investment for themselves or their family.

I asked forest owners to recall their forest management actions over the last 10 years and most reported multiple actions on their land, though they varied by frequency and extent. Reported actions included precommercial thinning (removing non-saleable brush, small trees); commercial thinning (selectively cutting trees as part of a timber sale); ground fuel removal (removing downed logs, branches, grass and shrubs mechanically or manually); ladder fuel removal (limbing trees or removing small trees growing next to large trees), pile burning (burning woody fuels in piles); chipping (using a machine to chip woody fuels and spreading them or using as biomass for fuel); and using prescribed fire. I then classified owners by emergent categories representing how they manage forests on their property, and how useful their management actions are in the context of adapting dry conifer forests to climate change, whether intentional or not (Table 4.3). I considered those who commercially thinned and used prescribed fire as the most active managers in the context of climate change adaptation because numerous studies show combining these treatments reduces tree mortality following wildfires in North America’s dry conifer forests (Raymond and Peterson 2005, Wimberly et al. 2009, Prichard et al. 2010).

Table 4.3. Emergent forest management categories.

Management category (# of owners)	Description
Inactive (6)	Have not yet carried out any forest management activities on their property.
Moderately active (20)	Performed some fuels management in the form of precommercial thinning, prescribed burns, manual ground fuel removal, chipping, and/or pile burning.
Very active (15)	Commercially thinned all or a portion of their property, and performed fuels management in the form of precommercial thinning, manual ground fuel removal, chipping, and/or pile burning.
Extremely active (9)	Commercially thinned all or a portion of their property, performed fuels management in the form of precommercial thinning, manual ground fuel removal, chipping, and/or pile burning, and conducted prescribed burns.

I did not observe connections between an individuals’ perceived risk of wildfire on their own property and their level of active forest management. Some of the most active managers viewed their wildfire risk as low because of their management, while others viewed risk as always high in their region. Some forest owners were very concerned about wildfires on adjacent public lands spreading onto their property, which they viewed as poorly managed, while others described how topographic, vegetation, and local weather conditions made risk variable over time.

Below, I first present the intentional climate-adaptive forest management participants reported, followed by the barriers I identified. Illustrative quotes employ the respondents’ ID codes, corresponding to their county of residence (C: Crook; WH: Wheeler; G: Grant; W: Wallowa).

4.4.2 Climate-adaptive management actions

Only two out of the 50 forest owners interviewed said they were intentionally managing their forest to adapt to climate change through anticipatory actions. The first of these two individuals expressed concern over the effects of climate change on area forests and were carrying out multiple forest management actions in response, putting them in the emergent category of "extremely active manager". They raised the issue unprompted following the first interview question about management goals and priorities:

> G14: "Our goal is to maintain our property as a forested site in the face of climate change. . . I think that if areas in the southern Blue Mountains aren't managed it will be deforested —burned over and not reforested."

Participant G14's primary concern was that climate change is contributing to increasingly large stand-replacing wildfires, and that future warming and more frequent drought would reduce and eventually prevent post-fire seedling establishment and subsequent forest recovery. Therefore, they were taking comprehensive action to reduce the risk and impacts of wildfire on their property through thinning, limbing, prescribed burning, and annual manual fuel removal over their entire 40 acres (16 ha), with the exception of leaving some woody material as nurse sites for established seedlings (which I observed growing at low densities). They also purposely moved seed (cones) from their most vigorous ponderosa pine trees to sites they judged favorable for seedling establishment and growth. They also took steps to mitigate their contribution to climate change using on-site renewable energy generation. The second landowner who reported intentionally adapting to climate change focused on species diversity as a "no-regrets" management strategy:

G19: “Global warming is a problem . . . that’s one of the reasons we're trying to keep diversity [in our forests].”

Both of these landowners indicated that they perceived sufficient risk of negative impacts on their forests from climate change that they were intentionally managing them in specific ways. Besides these landowners, seven other participants suggested, or “envisaged” (van Gameren and Zaccai 2015) potential adaptation actions that they might implement in the future, which generally aligned with the recommendations from forest scientists (Table 4.1). These included suggestions to change species composition and/or maintain species diversity, as well as more intensive thinning, however all of these individuals took a reactive approach to adaptation. The following quotes demonstrate this reactive approach:

> W6: “Yes, climate change will stress forests and trees will become disease prone and will die. Overall it may have effects on species, which ones can live in certain areas. I'm keeping an eye on it but I haven't planned explicitly for it. I’m in watch and wait mode — [I have] talked to [Local Forester] about possibly needing to thin even more if drought happens.”

> G7: “I'll continue to manage to reduce wildfire risk, I’ll do what I’ve been doing. If I notice it getting drier and drier and drier, then yeah, I'll probably thin the inventory a bit.”

> C4: “…no one really knows what's going to happen, I haven't thought that far in advance —I would harvest if trees got too unhealthy.”

These forest owners perceived potential future risks, but uncertainty about local climate change impacts and a lack of perceived immediate harms appeared to underlie their reactive approach.

In order to understand the potential role of planning for the future in adaptive thinking, I also asked individuals about forest management plans (FMPs) as well how far into the future they planned their management actions. I then classified management plans by their quality in four emergent categories — comprehensive plans were those that were written down and included specific goals as well as a forest inventory (18 owners), while partial plans consisted of either an inventory or written goals (12 owners). Informal plans were those that individuals had "in their heads" (9 owners), and the rest had no management plan (10 owners). One landowner who co-managed their land with a relative was unsure of what type of plan they had.

Most reported taking forest management actions as the opportunity arose (24 owners), while others reported planning less than 10 years ahead (15 owners). Four planned 10-20 years into the future, and four reported planning greater than 20 years into the future (three others did not answer this question). Most of the individuals who planned more than 10 years into the future had comprehensive management plans, and most of those who suggested or implemented adaptation actions had comprehensive plans. However, only two forest owners overlapped between these groups, because several of the forest owners planning more than 10 years ahead did so with the goal of optimizing timber rotations and enhancing logging profits, and were not necessarily concerned about climate change.

4.4.3 Subjective barriers to adaptation

I identified several subjective barriers to adaptation. First, beliefs about the cause of climate change affected individual risk perceptions and subsequent motivation to take action. The vast majority agreed that climate change is occurring, but many (19 owners) said climate change is due to "natural cycles" (Figure 4.2a):

> G3: "I think there's some climate change, but I don't think it's human-caused. It's in a constant state of flux and there's not much I can do about it."

Some who believed the natural cycle explanation expressed hopes that the trajectory would change:

> C7: "There's a warming trend, but I hope it's a natural cycle and a short cycle."

Fewer thought it was due solely to anthropogenic activities (11 owners), and others attributed changes to a combination of anthropogenic activities and natural causes (9 owners) (Figure 4.2a):

> G1: “The pines are being stressed. *Diplodia* [a conifer disease] was not an issue before — I think I'll lose 50% of my seed trees in the next 5 years…The climate is changing, and some of it is a natural cycle and some of it is humans.”

When asked about perceived local environmental change, the three most common landowner observations were that it is becoming hotter and drier, and most commonly that winter snowfall has declined considerably over the years (Figure 4.2b).

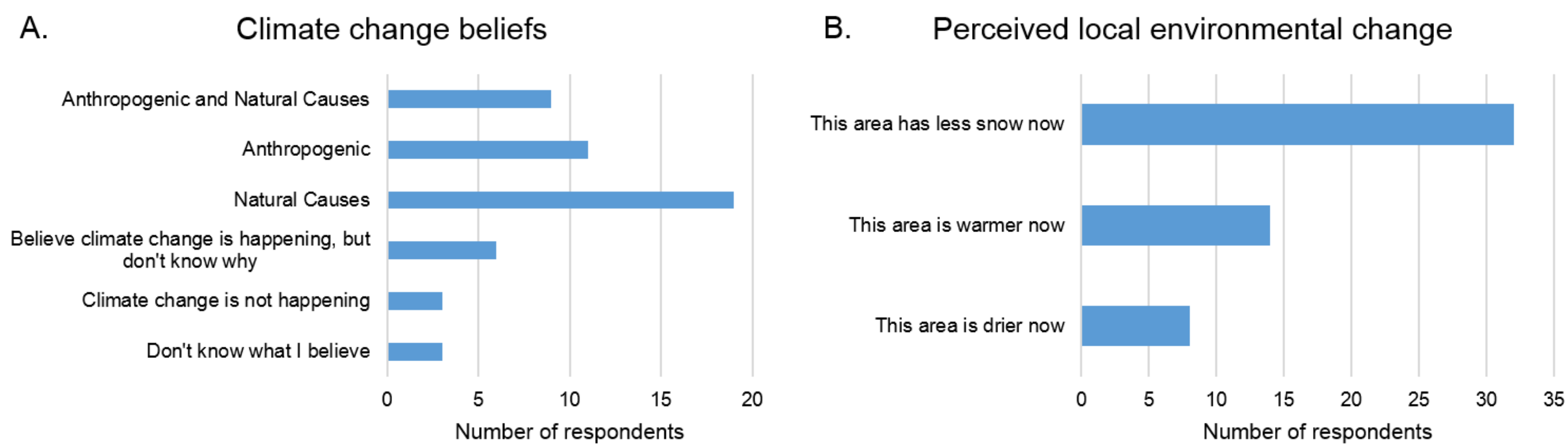

Figure 4.2. (A) Beliefs about climate change and its causes (B) and perceptions of local environmental change among non-industrial private forest owners (N = 50).

The nine individuals who suggested or implemented intentional climate-adaptive actions did not report perceiving local environmental changes differently than those who did not discuss adaptation, primarily because most people, regardless of their beliefs about climate change, reported declines in snowpack. However, individuals who suggested or implemented climate change adaptation strategies more often believed that climate change is occurring due to anthropogenic activities (6 out of 9) than the broader pool of participants (11 out of 50). Others who believed climate change is due to anthropogenic activities speculated on potential impacts, but expressed uncertainties about local impacts, ultimately stopping short of discussing potential personal adaptations:

> WH3: "I'm a personal believer in climate change. I'm not sure how it's playing out in our personal situation, but it would be interesting to see local climate data. Maybe less long cold spells? And that would result in more bugs in the forest."

> W12: "…looking at scientific predictions I don't know what the local impacts will be, it's a huge question. I don't know what the effects are yet."

Overall, uncertainty was a common theme, including uncertainty around the cause of climate change and its future trajectory as well as its local implications.

4.4.4 Structural barriers to adaptation

To understand structural barriers to adaptation, I asked forest owners if they faced challenges implementing management actions or if there were specific resources they needed to accomplish their management goals. Common themes included a need for light logging equipment and more grant or cost-share funding to support activities (Table 4.4). Others were systemic barriers including weak forest product markets and therefore no financial incentive for active management, and a dearth of social capital, including insufficient communication between adjacent property owners and a lack of affordable local labor (Table 4.4).

Table 4.4. Common management resource constraints expressed by forest owners that pose barriers to intentional and incidental adaptation.

Resource Need	Description
Education	Forest owners expressed a desire for more workshops specific to local ecosystems. Some felt that workshops were too basic or did not otherwise align with their interests. Others were aware of workshops but had not attended any yet, either because they had not made it a priority, or because of concerns over travel distance.
Equipment	Several owners expressed a desire for some kind of equipment sharing, rental or cooperative program for light logging equipment (e.g. compact feller buncher) and chippers, which may be too expensive for individual owners to buy and maintain.
Grants/Cost-share	Some owners said they knew where to apply for grants but had not done so yet, while others said they did not know where to find information on grants, and others said grants/cost-share programs needed to provide more dollars. Some also expressed reservations about grants because of the stipulations of certain grants.
Labor	Many interviewees are of retirement age and said that they needed help with labor. Several expressed concerns that hiring labor is too expensive, that there are not enough skilled forestry workers in their area, or had concerns about liability.
Better forest product markets	Several landowners said they needed a profit incentive to actively manage their forests, including a market for small-diameter timber or a chip market, and that log prices were too low to make commercial thinning economically viable. Specifically, with fewer mills in the region, several respondents said log hauling costs were prohibitive and undercut any income from timber sales.
Institutional capacity	A few landowners expressed a desire to work across parcel boundaries with neighbors on forest management, whether public or private neighbors.
Time	Many said they did not have time to do what needed to be done, or that they would get around to taking action eventually.

I compared the resource needs of forest owners in each of the emergent active management categories, and found that inactive or moderately active managers generally reported multiple resource needs, while very active and extremely active managers generally reported only one or two resource needs, if any. While this finding is intuitive, it reveals that addressing resource needs may facilitate active forest management, promoting both intentional and incidental adaptation. Some of the extremely active managers had learned how to successfully navigate grant and cost-share programs and used them to fund much of their management, while others owned sufficient timber or had the personal expertise and equipment (i.e., they were professional foresters themselves) to undertake forest thinning projects. Inactive and moderately active managers more commonly reported needing help with physical labor (either volunteer or paid) to help them carry out treatments.

I identified one final potential barrier to implementing climate-adaptive forest management, which was the perceived tradeoff between thinning and timber yields, as one participant explained:

> W11: "Less moisture means more stress on trees, more mortality, and higher burn risk. I want to leave things heavily stocked enough but ensure each tree is healthy. I'm worried I may be leaving things too close —but I want to save merchantable timber."

This perceived dilemma underscores the intersection of structural barriers to adaptation, such as the costs associated with both carrying out management actions and potentially sacrificing merchantable timber, with subjective barriers, exemplified by uncertainty regarding drought-stress effects on trees and therefore what stocking levels are best moving forward.

4.5 Discussion

Overall, my results demonstrate that intentional climate change adaptation generally has low salience among forest owners in eastern Oregon, and is much less common than in Europe, aligning with findings from focus groups performed in the western US five years prior to my interviews (Grotta et al. 2013). However, a large majority of forest owners are implementing incidentally adaptive actions including thinning and general fuels management in service of other goals, including timber growth and yield, wildlife habitat and wildfire risk mitigation. Additionally, by comparing one-fifth of forest owners who are either considering or implementing adaptation actions with those who are not, I identified multiple important subjective and structural barriers that constrain adaptation in different ways. Subjective barriers primarily barred intentional climate change adaptation, while structural barriers constrained both intentional and incidental adaptation. My findings contribute to arguments that the importance of subjective versus structural barriers depends on context, and that they interact to determine adaptation outcomes (Eisenack et al. 2014).

Most forest owners in this case study agreed that climate change is occurring. However, beliefs about the causes of climate change appeared to influence the ways in which acknowledgment of the phenomenon affected risk perceptions and motivations to adapt. Those who believed that climate change is occurring due to natural cycles were generally not confident about the future trajectory of climate change and its potential impacts, which likely reduced their level of concern and undercut motivations to take intentional adaptive actions. These individuals would likely be members of The Doubtful group of global warming's "Six Americas", a segmentation of the US population by Leiserowitz et al. (2009). Aligning with this typology, many of these respondents were male, white, older, and Republican.

In contrast, the seven other landowners who suggested adaptation actions predominantly identified as Democrats, or left-leaning Independents, despite also being older and predominantly men. Along with the two landowners who intentionally implemented climate-adaptive management, they believed the scientific consensus that anthropogenic emissions cause climate change. Landowner G14, who described their intentionally climate-adaptive forest management, had installed solar power and hot water heating to reduce their greenhouse gas emissions. They would likely fall into the Alarmed group of the Six Americas, while others who suggested potential adaptation actions align with the Concerned group (Leiserowitz et al. 2009). They possessed greater certainty that warming would continue in their local area, contributing to local environmental changes such as increasingly frequent wildfires, pests and disease, and drought stress in trees. In Figure 4.3, I illustrate the factors I observed impacting each stage of the adaptation process among private forest owners, including how climate change beliefs and concern operating in the understanding phase drive adaptation outcomes.

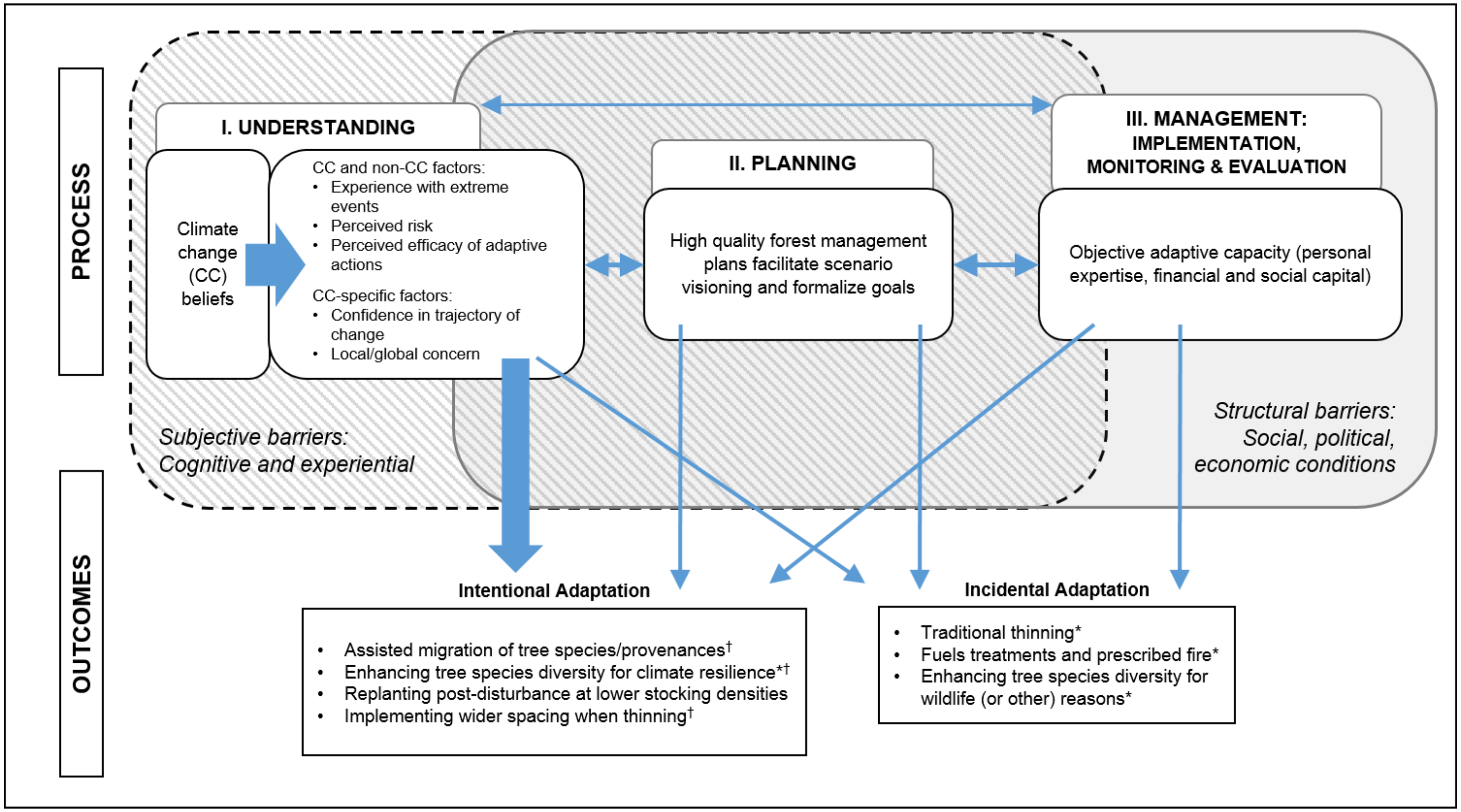

Figure 4.3. The adaptation process of non-industrial forest owners in eastern Oregon, with arrows indicating connections between process components and outcomes. I identified important factors influencing individual adaptive capacity in the understanding, planning, and management phases of adaptation. Subjective and structural barriers to adaptation may arise at each stage, but beliefs about climate change and their effects on risk perceptions may primarily determine whether forest owners engage in intentional climate change adaptation. Forest management planning and individual adaptive capacity affect both intentional and incidental adaptation outcomes. *Adaptation actions reportedly implemented or †suggested for future implementation by forest owners interviewed in this study; other listed adaptation outcomes are recommended by forest scientists but were not reported (see Table 4.1).

These findings align with those of a survey of Swedish private forest owners (Blennow and Persson 2009). That study found a strong, statistically significant association between strength of belief in climate change and taking steps toward adaptation. There has been a slow upward trend in US public acceptance of anthropogenic climate change, rising above 60% by late 2017 (Hamilton 2017) though it was closer to 50% when these interviews were conducted in 2015 (Hamilton et al. 2015b). The fraction of people who reject anthropogenic climate change remains sizable in the nation, and proportionately larger in eastern Oregon. This barrier is difficult to overcome because it is strongly tied to political and cultural identities, and may limit intentional climate change adaptation (Hamilton et al. 2015a).

I observed a great deal of uncertainty about local climate change impacts and the efficacy of adaptation actions among individual landowners. This uncertainty not only precluded some landowners from considering adaptation actions, but also contributed to a reactive "wait-and-see" approach to adaptation. While a reactive approach may be effective for addressing some impacts, such as drought stress impacts on individual trees, it may be maladaptive for reducing risks posed by increasingly frequent large wildfires and insect outbreaks, which may only be mitigated by more widespread reductions in tree densities or creating more heterogeneous tree coverage (Hessburg et al. 2015). This finding aligns with other studies finding forest owners view lack of access to easily interpreted information on local climate projections and adaptation options as a structural barrier to adaptation, which in turn impacts their subjective perception of personal adaptive capacity (Blennow and Persson 2009, Grotta et al. 2013, Lawrence and Marzano 2014, Sousa-Silva et al. 2016).

Forest owners who did not engage in widespread fuel reductions or other adaptive actions said they face multiple resource-based barriers to active forest management, impacting their adaptive capacity. These included insufficient time and financial resources and lack of access to equipment and labor. Removing these structural barriers may result in more climate-adaptive forest management, whether intentional or incidental (Figure 4.3).

Forest owners with timber production goals were some of the most active managers I observed, with many performing thinning and fuel removal both to reduce wildfire risk and increase growth and yield. While these incidental adaptations are beneficial, they may not be sufficient given projected climate impacts. Owners with timber production goals may have a conflict of interest between maximizing trees per acre and thinning to stocking levels that are more sustainable under hotter, drier conditions. A similar study in Belgium identified the same potential conflict (van Gameren and Zaccai 2015). Additionally, production-focused owners may prioritize fast-growing species and monocultures over future climate-adapted species or species diversification. Interview responses contained almost no discussion of planting trees using stock from hotter, drier regions, or intentionally favoring future climate-adapted species on specific sites. This perhaps should not be surprising, as assisted migration is controversial and still gradually gaining traction in industrial forestry contexts, with only a few assisted migration policies arising in countries like Canada (Klenk 2015). However, my findings contrast with studies from Europe where adaptation-focused experimentation with species compositions is reportedly common (van Gameren and Zaccai 2015, Sousa-Silva et al. 2016). This difference may partly reflect the low species diversity of western North American conifer forests.

4.5.1 Communication and policy recommendations

I found uncertainty surrounding projected local climate change as a key factor underlying forest owners' justification for reactive approaches to adaptation. However, downscaled projections for climate change impacts exist for the Blue Mountains (Halofsky and Peterson 2016) as well as many other forested regions globally, suggesting this is a knowledge transfer problem (Sousa-Silva et al. 2016). Scientists and practitioners must clearly explain uncertainties associated with projected local impacts, and articulate adaptation actions that are likely to be effective, as well as those that represent "no regrets" strategies. Numerous studies indicate that this type of information is best delivered by trusted community members through experiential learning activities, such as forestry site tours and deliberative workshops (Hobson and Niemeyer 2011, Raymond and Robinson 2013, Klein et al. 2014). Organizations and agencies could cultivate local champions who can help make conversations about climate change and adaptation more socially acceptable. Climate scientists could also present at forest management training days, or contribute clearly written, simple articles outlining likely local scenarios to local newspapers and resource management newsletters. Strong governance signals in the form of supportive policies or programs are also essential for fostering public responses to climate change (Hobson and Niemeyer 2011).

At the same time, my work suggests that in regions where climate change discussions are politically charged and may lead to disengagement, it may also be beneficial to focus communication strategies on drivers of incidental adaptation such as wildfire risk. Incidental adaptation actions, while potentially insufficient on their own to ensure forest resilience long-term, will yield public benefits as the climate warms (Hartter et al. 2017).

I found that forest owners with comprehensive forest management plans often have longer planning horizons and more frequently suggested or implemented adaptation actions. The causality of this relationship may operate in both directions (individual concern about climate change and interest in adaptation may encourage these landowners to develop management plans) (Figure 4.3). However, the process of developing a management plan, especially if done with input from an outside expert, may encourage "visioning" of future conditions and consideration of the long-term implications of management decisions. Other studies indicate landowners with written forest management plans are more likely to engage in silvicultural activities and express interest in ecosystem management (Creighton et al. 2002, Joshi and Arano 2009). Therefore, forest management plans, land stewardship plans, or other private land management planning processes may be an effective intervention point for increasing private landowner engagement with adaptation.

Management plans with long planning horizons may increase the salience of projected climate change impacts. Most climate change projections are 30, 50 or 100 years in the future — timelines that may not be relevant to many private forest owners as shown elsewhere (Grotta et al. 2013, Bissonnette et al. 2017). In this study few forest owners planned greater than 10 years ahead. Oregon Forest Management Plan Guidelines (2013) currently recommend private forest owners write management plans with a ten-year planning horizon. Extending this recommended horizon may encourage landowners to think about longer-term processes, including climate change impacts and adaptation. The USDA Forest Service's Climate Change Response Framework recognizes this, and has developed a detailed Adaptation Workbook for forest owners and managers in the US Northeast and Midwest (Swanston et al. 2016, Ontl et al. 2018), which provides adaptation recommendations and encourages both short (<10 years) and long-term management plans (>30 years). Such a workbook would likely be beneficial for non-industrial private forest owners in the western US.

Overall, my work suggests several specific recommendations for organizations and agencies supporting climate-adaptive forest management among non-industrial private forest owners:

- Ecoregionally-relevant education on climate change adaptation actions, including clear recommendations for tree density management and species composition.
- Incentives to complete management plans with multi-decadal time horizons.
- Cooperative or rental programs for light logging machinery and other equipment to lower management costs and facilitate active management.
- Accessible grant and cost-share programs to improve affordability of climate-adaptive forest management where markets for wood products are depressed.
- Multi-parcel projects through collaborative structures or cooperative agreements that allow landowners to pool timber, financial resources, equipment and labor, including public-private partnerships to increase adaptation efficiencies.

Some of these recommendations are likely relevant to climate change adaptation on private lands in a variety of socioecological systems. These include managing to reduce flood and erosion risks, enhancing water availability in arid regions, and maintaining wildlife habitat.

4.5.2 Limitations

I recognize that by conducting these interviews over the span of a single summer in 2015, my findings may not reflect evolving views among non-industrial private forest owners. Near the end of that summer eastern Oregon experienced several large, high-severity wildfires, in which over 30 families lost their homes. These events could have changed levels of concern about climate change, though in affiliated telephone interviews I conducted I saw no change between the fall of 2014 and 2015 in beliefs about the cause of climate change (Hamilton, unpublished data).

The forest owners I interviewed are also likely far more active than the average forest owner in eastern Oregon. Most volunteered to participate or were referred by another landowner, indicating they are at least somewhat engaged in their community and interested in forest management. Approximately half of the participants had commercially thinned a portion of their property. A large mail survey of forest owners in this region and the nearby eastern Cascade Mountains found 36% reported thinning with mechanized equipment, and 29% reported harvesting timber for profit between 2003 and 2008 (Fischer and Charnley 2012), indicating that my participants are likely more active than typical forest owners. I also asked participants about their participation in forestry workshops, and roughly half reported attending workshops held by Oregon State University Extension (Boag, unpublished data), aligning with findings from another mail survey previously conducted in this region (Hartter et al. 2015). Random sample telephone or mail surveys and inferential statistics would be necessary to evaluate the connections I identified in this qualitative study between climate change beliefs, forest management planning, resource needs, and adaptation outcomes.

4.6 Conclusion

This study contributes to a growing body of research on opportunities for and barriers to climate change adaptation among private forest owners, and individual decision makers more broadly. The forestry community should exploit synergies between managing forests for improved growth and yield, mitigating wildfire and other existing risks, and climate-adaptive forest management. In western North America, addressing existing resource needs for fuel reductions and density management will contribute to this synergistic approach. However, scientists, university extension services and agencies supporting private landowners should also continue to advance conversations about intentional climate change adaptation, while being sensitive to political polarization on the issue. Incidental and/or reactive adaptation may be an effective strategy for adapting to some climate change impacts including sporadic drought (i.e., thin when trees start to look stressed), however it may be inefficient in terms of long term social, political and economic costs. Believing anthropogenic climate change is occurring is a precursor to accepting that – unless emissions are dramatically reduced – local warming and its associated impacts on forests will continue through the 21st century. Confidence in the trajectory of change is necessary to begin conversations about thinning forests below historical stocking levels or favoring certain tree species.

This logic extends to decisions that may be made by groups of forest owners. Some very new groups of non-industrial private forest owners such as the Ritter Land Management Team (Ritter LMT 2017) in eastern Oregon are emerging to pool resources and carry out multi-parcel forest management projects across private lands. They are also interfacing with state and federal agencies to engage in an “all lands approach” to forest management (Charnley et al. 2017). In forestry and other sectors, understanding barriers to adaptation among individual landowners will improve existing and potential adaptation strategies, enhancing the management of socioecological systems as the climate changes.

CHAPTER V

CONCLUSION

This dissertation examined eastern Oregon conifer forests as a dynamic socioecological system impacted by climate change, wildfire, and management decisions. I took a multi-disciplinary approach to understanding forest vulnerability and the adaptive capacity of forest managers using ecological modeling, field data collection and analysis, and qualitative social science. My work is unique in that it recognizes the region-specific nature of climate change impacts and adaptation options, and in addition to identifying local vulnerabilities, also investigates forest owner needs for climate-smart management. I found that in line with trends across western North America, the resilience of forests in the Blue Mountains is challenged by interactions between a warming climate and more frequent large wildfires. These trends are expected to continue and be amplified in the future unless global emissions are sharply and rapidly curtailed. Changing climate and wildfire regimes present diverse forest management challenges in eastern Oregon, which is an area with low public engagement on climate change.

In Chapter 2 I used a forest landscape simulation model run for 90 years to understand the effects of "worst case scenario" climate change (RCP 8.5), moderate climate change (RCP 4.5), contemporary and high wildfire activity on tree species establishment and growth. I analyzed differences between scenario outcomes by land ownership group in order to identify landowner-specific climate change adaptation priorities. I found species establishment probabilities declined from 2015-2100 under both RCP 4.5 and 8.5 for most species except ponderosa pine and Douglas-fir, while biomass and abundance increased for most species over the study area. Ponderosa pine and Douglas-fir experienced the largest increases in biomass, likely due to relatively high tolerance to drought and wildfire. In contrast, moist and cold mixed-conifer species, including Engelmann spruce, western larch and subalpine fir, all experienced biomass and abundance declines by 2100 under high fire activity and/or climate change scenarios.

In Chapter 3 I surveyed 184 sites across eight burned areas 15-21 years post-fire to understand how topography, climate, and post-fire legacies influence post-fire juvenile conifer abundance. One-third of sites contained no juvenile conifers 15-21 years post-fire, potentially indicating regeneration limitation or failure. These sites were primarily in ponderosa pine, Douglas-fir, and grand fir – dominated plant associations on steep, south-facing slopes below 4,000 feet elevation, aligning with findings from Chapter 2 projecting lower species establishment probabilities for Douglas-fir and ponderosa pine in the hottest, driest sites. However, juvenile conifer abundance in some north-facing sites near seed sources amply achieved recommended stocking levels. The most important variables predicting juvenile conifer presence included site heat load, overstory density, and distance to seed source, emphasizing the importance of topography and burn patterns in determining post-fire forest structure. Saplings were more likely to be present above 1500-m (~5,000 feet) where they were observed in very high densities in some cases. Additionally, I found that drought indices for the first three years post-fire were not an important predictor of conifer seedling or sapling presence. These results suggest forests in cool microsites that experience low to moderate burn severities are resilient to wildfire and drought, despite projections from Chapter 2 for declining species establishment, biomass and abundance for some tree species over the 21st century. Ongoing monitoring will be necessary to detect changes in species vulnerability as the climate warms, and this study provides a baseline dataset with which to compare potential future changes.

Finally, in Chapter 4 I gauged the capacity for climate change adaptation among private forest owners in eastern Oregon by interviewing 50 private landowners about their forest management activities, management plans, and perceptions and beliefs regarding climate change and wildfire. Intentional climate change adaptation had low salience among landowners, but the majority were active forest managers motivated by other goals, notably wildfire risk mitigation. The few landowners who implemented or intended to implement adaptation actions generally believed anthropogenic climate change is occurring. They also more frequently had formalized forest management plans compared with those not engaged in intentional adaptation. My work emphasizes that belief in anthropogenic climate change is necessary for landowners to consider intentional climate-adaptive forest management actions such as planting "future-adapted" species or genotypes. I encourage additional research on climate change beliefs and forest management in the western US given the small number of intentional adapters I identified.

Despite a lack of engagement with climate change, many forest owners in eastern Oregon implemented or planned to implement specific forest management activities motivated by wildfire risk mitigation and other factors that were nonetheless beneficial from an adaptation perspective, such as fuels reductions through commercial or precommercial thinning. Additionally, those intentionally and incidentally adapting to climate change faced many of the same barriers to implementation, including a need for eastern Oregon-specific forestry education, cooperative or rental programs for light logging equipment, more accessible and abundant grant and cost-share programs, as well as opportunities to engage in collaborative projects with other landowners or managers to pool timber, financial and equipment resources. In other words, regardless of each landowner's climate change beliefs, these private forest owners all experience educational, financial, and operational barriers to sustainable forest management in eastern Oregon.

5.1 Recommendations for climate-smart forest management

My research indicates several management priorities for forest owners and managers in the Blue Mountains, in addition to recommendations for university extension and other supporting organizations. At the site level, large trees should be protected for their resilience to wildfire and their role as an important seed source. When wildfire or harvesting does occur it may provide opportunities for experimentation with novel species assemblages and densities. In particular, if wildfire creates deforested patches lacking a seed source within ~100-m, replanting will likely be necessary to maintain forest cover, but replanting should be considered in the context of future conditions. This may mean replanting at lower-than-historic stocking densities to reduce future moisture stress, or even leaving some areas unplanted to restore heterogeneity to the forest landscape.

More broadly, private forest owners in eastern Oregon manage a wide diversity of forest types which may be vulnerable in different ways to climate-wildfire interactions, and therefore require different educational resources and management approaches. Both private and National Forest land in eastern Oregon contains a mix of dry, moist, and cold mixed-conifer forest types. In dry conifer forests, fuel treatments, prescribed burning, and more frequent wildland fire use will restore these forests to their natural range of variability and enhance resilience to climate change and wildfire. In contrast, cold mixed-conifer sites may actually benefit from ongoing fire suppression to maintain refugia for vulnerable species as temperatures warm. While this strategy cannot be successful indefinitely if temperatures continue to rise, it offers a potential resistance-oriented management approach. Refugia will most effectively preserve vulnerable forest types if they are replicated and connected across forest landscapes, necessitating cross-landownership collaboration.

In order to engage private forest owners in active forest management, including climate-smart forest management, existing support programs funded by USDA and state agencies must be strengthened and expanded. Findings from the National Woodland Owner survey indicate that landowners receiving assistance in the form of advice, management planning, and cost-share programs are two to three times more likely to implement wildfire hazard reduction and tree planting than their unassisted counterparts (Butler et al. 2014). My findings corroborate this research, showing that forest owners with detailed management plans are more likely to take climate-adaptive management actions. From a communications perspective, wildfire is the most commonly cited management concern among Oregon family forest owners after trespassing, with over 90% concerned (Butler and Butler 2016). Therefore, framing climate-adaptive forest management actions in the context of wildfire mitigation can be a communication strategy for university extension, state and federal staff who support private forest owners in regions where climate change is politicized.

However, some management discussions require invoking climate change. For example, my projections suggest that species establishment declines may occur in the coming decades both for ponderosa pine and Douglas-fir on hot and dry sites, as well as moist mixed-conifer species at high elevations. In response, landowners and managers will need to experiment with planting alternative provenances, genotypes, or species in order to maintain certain forest types, particularly on the most moisture-limited sites. Engaging with species experimentation and related questions around novel conditions and ecosystem states requires people accept that, whether anthropogenically-caused or not from their perspective, the warming and drying trend in their region will continue in the coming decades. I call on agency and support staff to open dialogues with the public about climate change and forest management to allow some of these discussions to begin. These dialogues will be challenging, but they should be respectful of dissenting opinions and rooted in data on the warming and drying trends already observed locally, which can be used as a springboard for conversations about ongoing change.

5.2 Key directions for future research

Given that understanding climate change impacts and devising adaptation approaches represents a “moving target”, monitoring will become increasingly important for detecting ecological thresholds and state transitions. My post-fire surveys of juvenile conifer abundance can be used as baseline data to assess successional trajectories through time, or investigate the impacts of successive wildfires if some of my sites burn in future years. In addition to region-specific observational studies, field experiments are needed to understand where and when sites may become unsuitable for certain species and more suitable for others. Field experiments may involve environmental manipulations using materials to shade, warm, or dry seedlings to gain a more nuanced understanding of the climatic determinants of growth. Such work is particularly important because forest landscape models are only as good as the empirical data used to parameterize and calibrate them, which is ideally sources from region-specific experimental or observational data. To expand the potential for regional-scale forest simulations we also need continued investments in forest inventory data collection, in addition to independent inventory datasets for model validation.

5.3 Closing thoughts

This dissertation evaluated changes in forest composition under a range of future climate and wildfire scenarios, assessed factors limiting juvenile conifer establishment to understand how climate change may impact establishment dynamics, and explored the adaptive capacity of local private forest owners. Overall, climate-adaptive forest management requires region-specific adaptive management based on experimentation. This experimentation should be replicated across land ownerships and requires expanding regional dialogues around local climate change impacts. Failure is guaranteed, but only through failure will we develop innovative sustainable management strategies for changing forests that succeed in maintaining their economic, ecological, and social values.

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Appendix A: LANDIS-II Model Input Parameters and Calibration

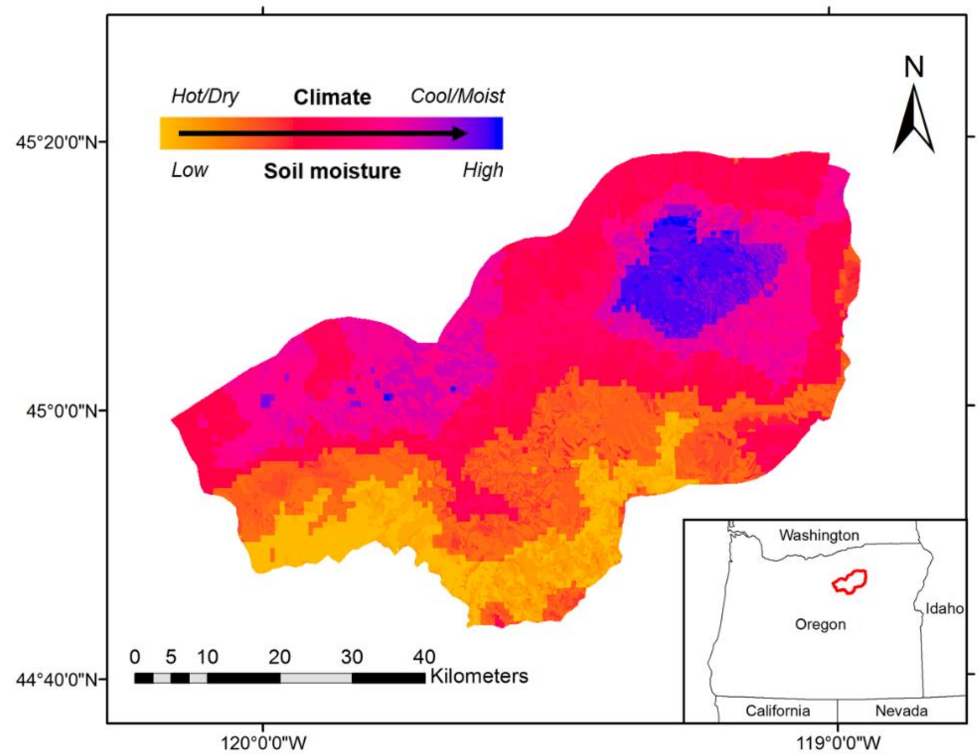

Figure A1. Study area showing land management and moisture gradient ranging from hot/dry climate and low soil moisture to cool/moist climate and high soil moisture. This gradient is represented by 25 ecoregions in LANDIS-II simulations.

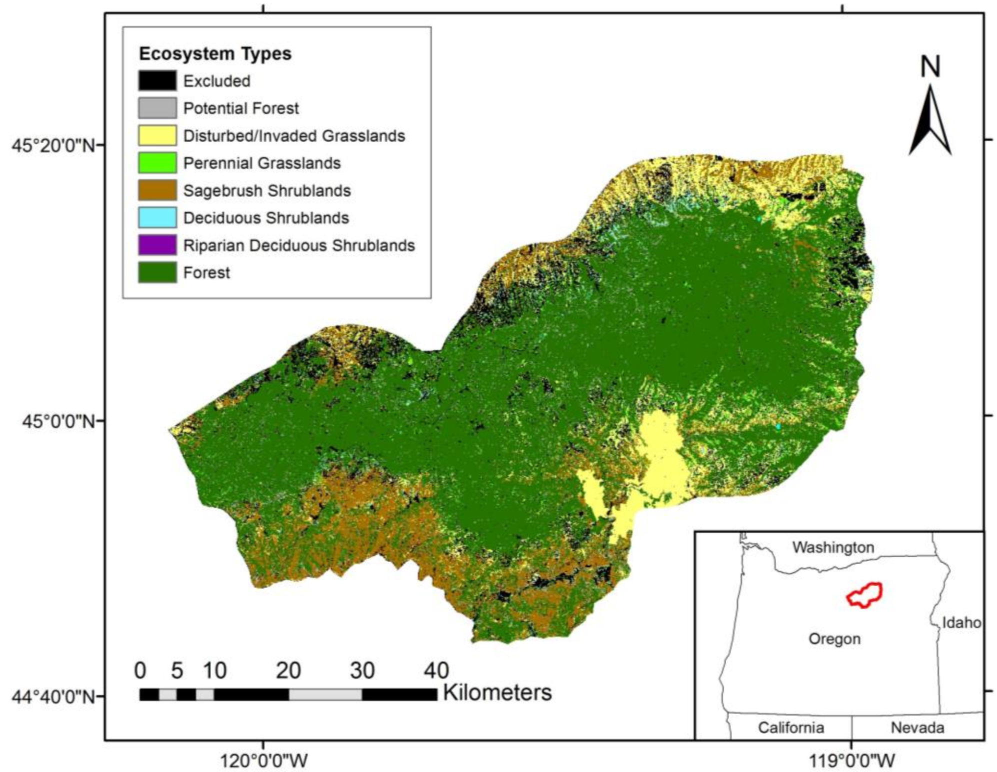

Figure A2. Initial ecosystem types within study area based on LEMMA GNN reclassification. The large area of disturbed/invaded grasslands in the south-central portion of the study area reflects plant communities following the 2001 Monument (Mallory) wildfire complex.

Table A1. Species life history parameters (Cassell 2018).

LandisData		“Species”									
>>			Sexual	Shade	Fire	Seed Disp Dist		Veg	Sprout	Age	Post-Fire
>>	Name	Long	Maturit	Tol.	Tol.	Effecti	Maximum	Rep P	Min	Max	Regen
>>	----	-------	-----	-----	----	------	-------	-----	----	-----	--------
	abiegran	300	20	4	3	30	300	0	0	0	none
	abielasi	200	20	5	1	30	80	0	0	0	none
	cercledi	300	10	2	2	30	400	0.2	1	120	resprout
	juniocci	1000	50	2	3	2	30	0	0	0	none
	lariocci	700	25	1	5	100	250	0	0	0	none
	pinucont	200	5	2	2	30	300	0	0	0	none
	piceenge	400	30	4	1	90	180	0	0	0	none
	pinupond	600	7	1	4	30	160	0	0	0	none
	pseumenz	300	15	2	3	80	240	0	0	0	none
	toleresp	60	3	3	1	30	1000	0.85	5	50	resprout
	intoresp	60	3	2	1	30	500	0.85	5	50	resprout
	nonnseed	80	5	2	1	30	250	0	0	0	none
	fixnresp	80	5	1	1	20	250	0.75	5	70	resprout
	natvgrss	100	1	4	1	1000	5000	1.0	0	100	resprout
	invsgrss	100	1	4	1	1000	5000	1.0	0	100	resprout

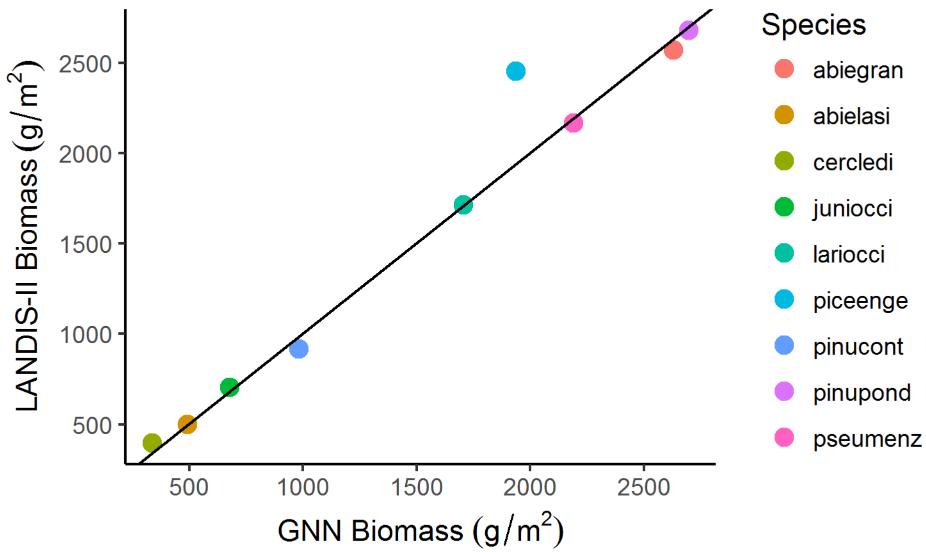

Figure A3. Final LANDIS-II year 0 biomass values following model spin-up vs. GNN biomass values after calibrating species growth.

Table A2. Dynamic Fire and Fuels System (DFFS) Dynamic Fire Region Table. Initial values were adopted from Cassell (2018). All fire regions were parameterized with the same fire behavior, and maximum duration values (**bold**) were 3,000 to simulate a contemporary fire size distribution, and 8,000 to simulate a “high fire” scenario. A. Boag determined these values through calibration runs.

LandisData "Dynamic Fire Region Table"

>> Fire Region Data Table

>>	Fire region		Fire Duration (minutes)				Seasonal Foliar Moisture Content (FMC) Data: Hi, Lo, and Proportion (Pr)									Other Ecorgion Data	
>>							Spring (Sp)			Summer (Su)			Fall (Fa)				
>> Year	Code	Name	Mu	Sigma	Min	Max	SpFMCLo	SpFMCHi	SpHiPr	SuFMCLo	SuFMCHi	SuFMCPr	FaFMCLo	FaFMCHi	FaHiPr	OpenFuel	NumFires
0	1	fire1	7	1.4	1	(**3000/8000**)	120	120	0.50	90	110	0.50	120	120	0.50	31	.4
0	2	fire2	7	1.4	1	(**3000/8000**)	120	120	0.50	90	110	0.50	120	120	0.50	31	.4
0	3	fire3	7	1.4	1	(**3000/8000**)	120	120	0.50	90	110	0.50	120	120	0.50	31	.4
0	4	fire4	7	1.4	1	(**3000/8000**)	120	120	0.50	90	110	0.50	120	120	0.50	31	.4
0	5	fire5	7	1.4	1	(**3000/8000**)	120	120	0.50	90	110	0.50	120	120	0.50	31	.4

The Dynamic Fire and Fuels System “Dynamic Fuels Classification” table was adopted from Cassell (2018).

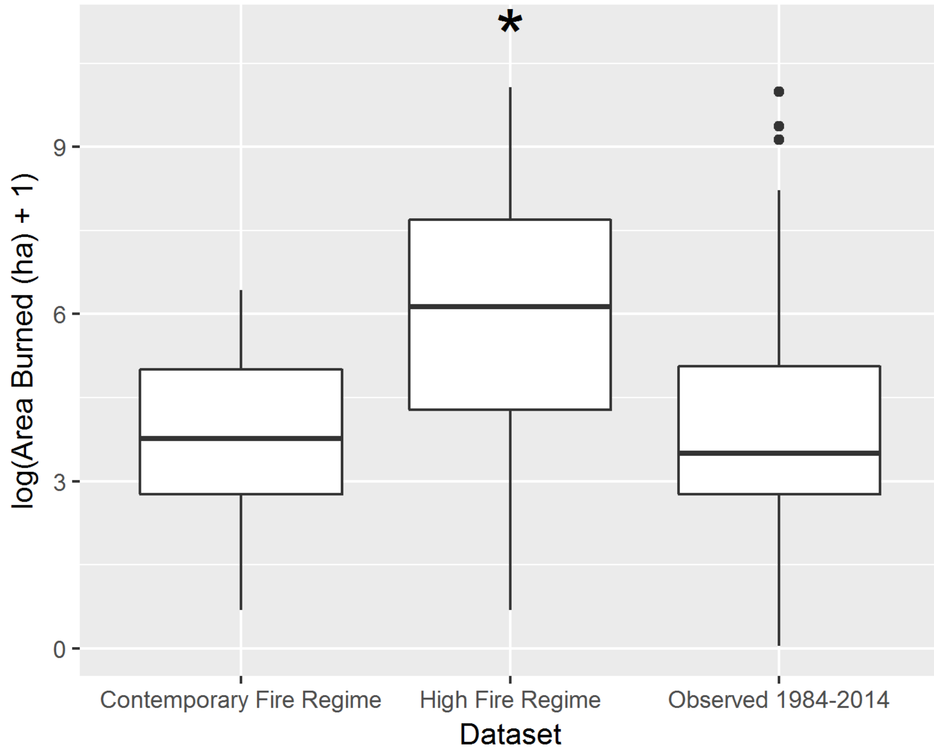

Figure A4. Wildfire scenario calibration for the Dynamic Fire and Fuels System (DFFS) extension. Fire size differs significantly between the High Fire Regime scenario and both the Contemporary Fire Regime scenario and fire sizes in the study area Observed 1984-2015 (Wilcoxon rank sum test, $p < 0.05$). The parameterized Contemporary Fire Regime scenario does not differ significantly from the historically observed data (Wilcoxon rank sum test, $p = 0.82$).

The following NECN Succession Extension v2.1 input tables were parameterized following Cassell (2018), whose parameters in turn are derived from existing LANDIS-II studies (Scheller et al. 2011, Loudermilk et al. 2014, Lucash et al. 2014, 2017, Creutzburg et al. 2016, 2017, Serra-Diaz et al. 2018), the Fire Effects Information System, and the USGS Climate-Vegetation Atlas (Thompson et al. 2015). Tables are available in B. Cassell's GitHub repository at: https://github.com/brookecassell/Project-Malheur_Fuels_Treatment

AvailableLightBiomass; LightEstablishmentTable; SpeciesParameters; FunctionalGroupParameters; FireReductionParameters

The following NECN Succession Extension v2.1 inputs were parameterized by A. Boag using data from the study area:

Table A3. Initial ecoregion parameters for soil organic matter carbon and nitrogen, and mineral nitrogen, parameterized for the study are using provisional SSURGO and STATSGO data.

InitialEcoregionParameters

>> Angela: I obtained SOM from ORNL website.

>>	SOM1	SOM1	SOM1	SOM1	SOM2	SOM2	SOM3	SOM3	Minrl
>>	C	N	C	N	C	N	C	N	N
>>	surf	surf	soil	soil					
eco101	24	2	48	5	1427	82	919	108	2
eco102	21	2	42	4	1244	71	801	94	2
eco103	22	2	44	4	1312	75	845	99	2
eco104	20	2	39	4	1165	67	750	88	2
eco105	22	2	45	4	1317	75	848	100	2
eco201	22	2	45	4	1327	76	854	101	2
eco202	22	2	44	4	1289	74	830	98	2
eco203	22	2	44	4	1307	75	842	99	2
eco204	22	2	44	4	1300	74	837	98	2
eco205	24	2	47	5	1390	79	895	105	2
eco301	25	2	49	5	1446	83	931	110	2
eco302	24	2	48	5	1427	82	919	108	2
eco303	25	3	50	5	1477	84	951	112	3
eco304	24	2	49	5	1445	83	931	109	2
eco305	24	2	47	5	1392	80	897	105	2
eco401	25	3	51	5	1504	86	969	114	3
eco402	26	3	52	5	1538	88	991	117	3
eco403	25	2	49	5	1460	83	940	111	3
eco404	25	3	51	5	1496	86	964	113	3
eco405	27	3	54	5	1589	91	1024	120	3
eco501	24	2	48	5	1411	81	909	107	2
eco502	24	2	48	5	1406	80	906	107	2
eco503	24	2	48	5	1403	80	904	106	2
eco504	23	2	47	5	1374	79	885	104	2
eco505	24	2	48	5	1412	81	909	107	2

Table A4. Further ecoregion parameters for soil characteristics and nitrogen cycling, parameterized for the study are using provisional SSURGO and STATSGO data. Nitrogen deposition parameters were calibrated using single-cell simulation runs by comparing simulated deposition values to empirical data from the National Atmospheric Deposition Program.

EcoregionParameters

>> Calculated clay, sand, field cap and wilt point values based on provisional SSURGO data and aggregation instructions from NRCS.

>>	Soil	Percent	Percent	Field	Wilt	StormF	BaseF	Drain	Atmos	Atmos	Lat-	Decay	Decay	Decay	Decay	Denitrif	
>>	Depth	Clay	Sand	Cap	Point	Fract	Fract		N	N	itude	Rate	Rate	Rate	Rate		
>> cm	frac	frac						slope	inter		Surf	SOM1	SOM2	SOM3			
eco101	100	0.09	0.26	0.05	0.02	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	<< for all ecoregions, StormF Frac -> Denitrif are constants from Pep's.
eco102	100	0.25	0.25	0.16	0.10	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco103	100	0.24	0.34	0.20	0.11	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco104	100	0.21	0.36	0.20	0.10	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco105	100	0.30	0.23	0.26	0.15	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco201	100	0.14	0.17	0.07	0.04	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco202	100	0.19	0.21	0.14	0.07	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco203	100	0.18	0.24	0.17	0.09	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco204	100	0.22	0.26	0.23	0.11	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco205	100	0.26	0.25	0.23	0.13	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco301	100	0.16	0.17	0.10	0.05	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco302	100	0.21	0.24	0.16	0.08	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco303	100	0.19	0.22	0.18	0.09	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco304	100	0.20	0.22	0.23	0.11	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco305	100	0.16	0.27	0.22	0.09	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco401	100	0.16	0.17	0.10	0.05	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco402	100	0.19	0.26	0.15	0.07	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco403	100	0.17	0.21	0.16	0.08	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco404	100	0.18	0.23	0.23	0.10	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco405	100	0.16	0.30	0.23	0.09	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco501	100	0.14	0.18	0.11	0.05	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco502	100	0.22	0.29	0.19	0.09	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco503	100	0.15	0.29	0.17	0.07	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco504	100	0.16	0.27	0.22	0.08	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	
eco505	100	0.17	0.28	0.23	0.10	0.2	0.2	1	0.00125	0.0055	45	0.8	0.8	0.03	0.0003	0.004	

Table A5. Maximum biomass parameters for species within each ecoregion. Initial values were 95th percentile biomass per species from the tree-level GNN data. Some values were changed during spin-up calibration runs to better reproduce observed total biomass values in the GNN data.

MaxBiomass << (g Biomass / m2)

>> Species Ecoregions

>> --------------------------
>> Angela: All values are 95th percentile total biomass (g/m2) per species across all GNN FCIDs, rounded to the nearest 100. Shrub and grass values are from Pep/Brooke.

	eco101	eco102	eco103	eco104	eco105	eco201	eco202	eco203	eco204	eco205	eco301	eco302	eco303	eco304	eco305	eco401
abiegran	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500	10500
abielasi	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000
cercledi	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100
juniocci	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
lariocci	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700
piceenge	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000
pinucont	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650
pinupond	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800
pseumenz	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300
toleresp	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
intoresp	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
nonnseed	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
fixnresp	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
natvgrss	700	700	700	700	700	700	700	700	700	700	700	700	700	700	700	700
invsgrss	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000

>> MaxBiomass cont'd

	eco402	eco403	eco404	eco405	eco501	eco502	eco503	eco504	eco505	
abiegran	10500	10500	10500	10500	10500	10500	10500	10500	10500	
abielasi	8000	8000	8000	8000	8000	8000	8000	8000	8000	8000
cercledi	2100	2100	2100	2100	2100	2100	2100	2100	2100	2100
juniocci	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
lariocci	6700	6700	6700	6700	6700	6700	6700	6700	6700	6700
piceenge	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000
pinucont	4650	4650	4650	4650	4650	4650	4650	4650	4650	4650
pinupond	8800	8800	8800	8800	8800	8800	8800	8800	8800	8800

pseumenz	7300	7300	7300	7300	7300	7300	7300	7300	7300	7300
toleresp	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
intoresp	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
nonnseed	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
fixnresp	2000	2000	2000	2000	2000	2000	2000	2000	2000	2000
natvgrss	700	700	700	700	700	700	700	700	700	700
invsgrss	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000

Table A6. Aboveground Maximum Net Primary Productivity (ANPP) parameters for species within each ecoregion. Initial values were adopted from existing research (Cassell 2018), and some values were modified during spin-up calibration runs to better reproduce observed total biomass values from the GNN data. Values for shrub and grass functional groups were also adopted from existing research (Cassell 2018 via Hansen et al. 2000).

MonthlyMaxNPP << PRDX(3) from Century 4.0 (g aboveground Biomass / m2/mo.)

>> Species Ecoregions
>> -------- ------------------
>> From Brooke

	eco101	eco102	eco103	eco104	eco105	eco201	eco202	eco203	eco204	eco205	eco301	eco302	eco303	eco304	eco305	eco401
abiegran	110	110	110	110	110	110	110	110	110	110	110	110	110	110	110	110
abielasi	900	900	900	900	900	900	900	900	900	900	900	900	900	900	900	900
cercledi	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000
juniocci	110	110	110	110	110	110	110	110	110	110	110	110	110	110	110	110
lariocci	250	250	250	250	250	250	250	250	250	250	250	250	250	250	250	250
piceenge	180	180	180	180	180	180	180	180	180	180	180	180	180	180	180	180
pinucont	900	900	900	900	900	900	900	900	900	900	900	900	900	900	900	900
pinupond	115	115	115	115	115	115	115	115	115	115	115	115	115	115	115	115
pseumenz	170	170	170	170	170	170	170	170	170	170	170	170	170	170	170	170
toleresp	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35
intoresp	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35
nonnseed	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35
fixnresp	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35
natvgrss	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35
invsgrss	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35

>> MaxANPP cont'd

		eco402	eco403	eco404	eco405	eco501	eco502	eco503	eco504	eco505
abiegran		110	110	110	110	110	110	110	110	110
abielasi		900	900	900	900	900	900	900	900	900
cercledi		1000	1000	1000	1000	1000	1000	1000	1000	1000
juniocci		110	110	110	110	110	110	110	110	110
lariocci		250	250	250	250	250	250	250	250	250
piceenge		180	180	180	180	180	180	180	180	180
pinucont		900	900	900	900	900	900	900	900	900
pinupond		115	115	115	115	115	115	115	115	115
pseumenz		170	170	170	170	170	170	170	170	170
toleresp		35	35	35	35	35	35	35	35	35
intoresp		35	35	35	35	35	35	35	35	35
nonnseed		35	35	35	35	35	35	35	35	35
fixnresp	35	35	35	35	35	35	35	35	35	35
natvgrss	35	35	35	35	35	35	35	35	35	35
invsgrss	35	35	35	35	35	35	35	35	35	35

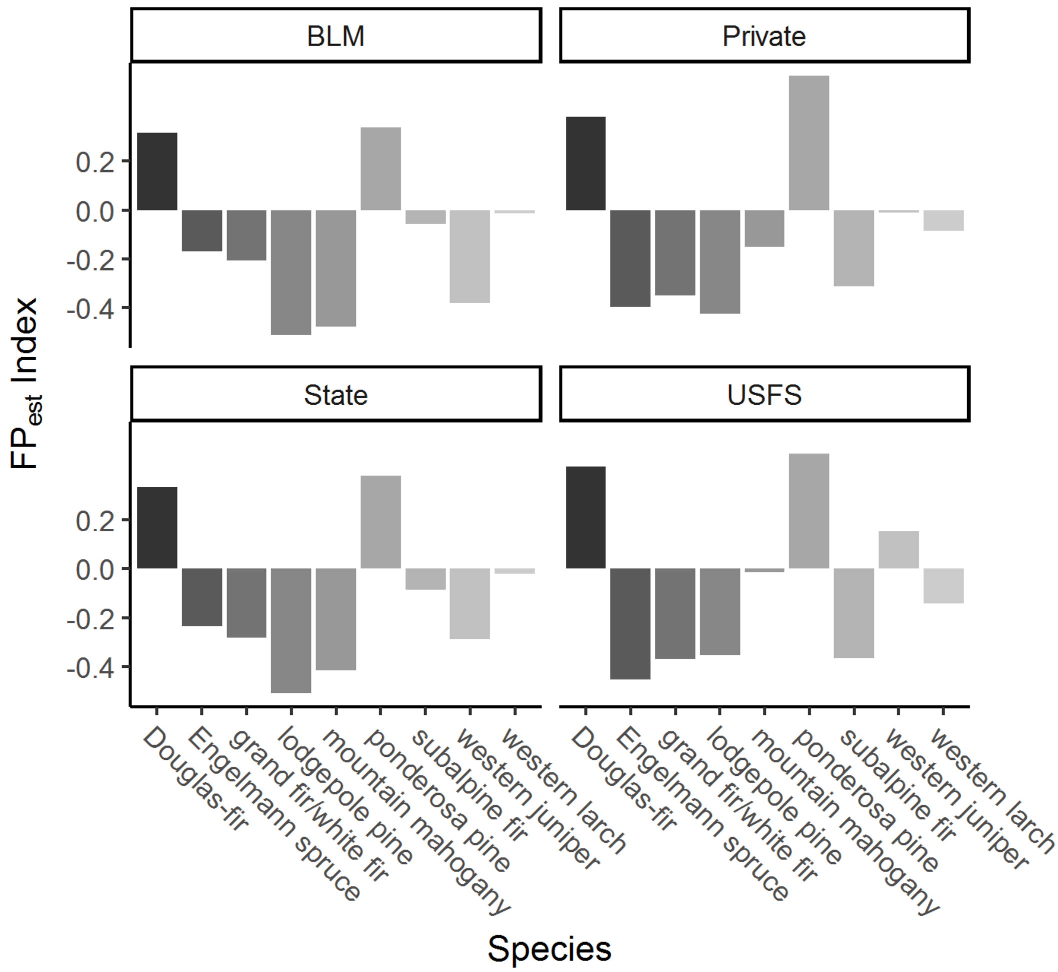

Figure A5. Significant changes in species establishment probabilities for 2010-2100 under RCP 4.5 by land ownership, represented by the Future Probability of Establishment Index (FP_{est} Index). The magnitude of the FP_{est} Index reflects significant projected changes (negative values = decline; positive = increase) in species establishment probabilities as well as the percentage of the landowner group's land area in which the changes are projected to occur.

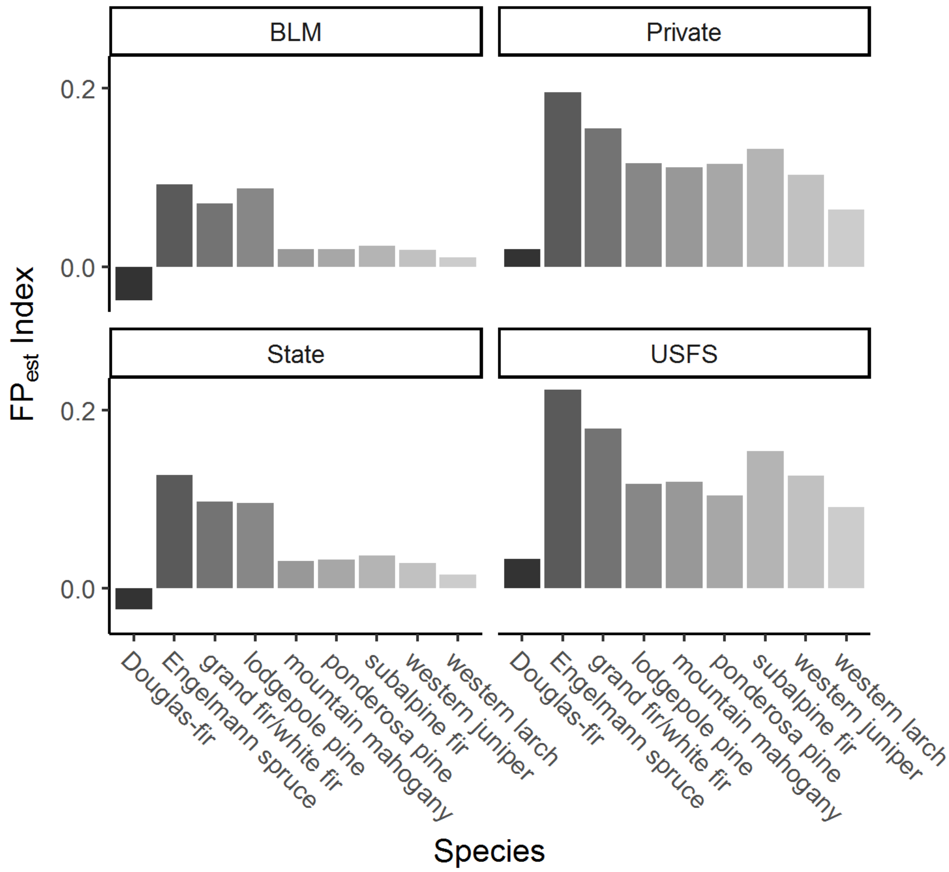

Figure A6. Significant changes in species establishment probabilities for 2010-2100 under current climate by land ownership, represented by the Future Probability of Establishment Index (FP_{est} Index). The magnitude of the FP_{est} Index reflects significant projected changes (negative values = decline; positive = increase) in species establishment probabilities as well as the percentage of the landowner group's land area in which the changes are projected to occur

Appendix B: Post-fire Regeneration Supplementary Data

Table B1. Suggested stocking levels for plant associations present at sites surveyed based on Table 2 in Powell 1999 (in turn based on Cochran et al. 1994). The Lower Limit of the Management Zone (LLMZ) represents the approximately 50% of full stocking.

Ecoclass	Plant Associations Present at Sites	Name	Lowest LLMZ (trees acre^{-1})	Lowest LLMZ (trees hectare^{-1})
CDG111	PSME/CAGE2	Douglas-fir/elk sedge	58	143
CDG121	PSME/CARU	Douglas-fir/pinegrass	82	203
CDS611	PSME/HODI	Douglas-fir/oceanspray	113	279
CDS622	PSME/SYAL	Douglas-fir/common snowberry	101	250
CDS711	PSME/PHMA5	Douglas-fir/mallow ninebark	112	277
CDS812	PSME/VAME	Douglas-fir/big huckleberry	64	158
CES131	ABLA/CLUN2	subalpine fir/queencup beadlily	205	506
CES311	ABLA/VAME	subalpine fir/big huckleberry	114	282
CJS41	JUOC/CELE3/ FEID-AGSP	western juniper/mountain mahogany/Idaho fescue-bluebunch wheatgrass	Not available	Not available
CLS416	PICO/CARU	lodgepole pine/pinegrass	Not available	Not available
CLS417	PICO(ABGR)/ VASC/CARU	lodgepole pine(grand fir)/grouse huckleberry/pinegrass	68	168
CLS513	PICO(ABGR)/ VAME	lodgepole pine(grand fir)/big huckleberry	93	230
CPG111	PIPO/AGSP	ponderosa pine/bluebunch wheatgrass	26	64
CPG112	PIPO/FEID	ponderosa pine/Idaho fescue	42	104
CPG222	PIPO/CAGE2	ponderosa pine/elk sedge	55	136
CPM111	PIPO/ELGL	ponderosa pine/blue wildrye	Not available	Not available
CPS522	PIPO/SYAL	ponderosa pine/common snowberry	146	361
CWF311	ABGR/LIBO3	grand fir/twinflower	108	267
CWF312	ABGR/LIBO3	grand fir/twinflower	108	267
CWF421	ABGR/CLUN2	grand fir/queencup beadlily	Not available	Not available
CWG111	ABGR/CAGE2	grand fir/elk sedge	73	180
CWG112	ABGR/CARU	grand fir/pinegrass	103	254
CWG113	ABGR/CARU	grand fir/pinegrass	103	254
CWS211	ABGR/VAME	grand fir/big huckleberry	93	230
CWS531	ABCO-ABGR/HODI	white fir–grand fir/oceanspray	Not available	Not available
CWS811	ABGR/VASC	grand fir/grouse huckleberry	68	168
CWS912	ABGR/ACGL	grand fir/Rocky Mountain maple	121	299
GB41	AGSP-POSA12	bluebunch wheatgrass-Sandberg’s bluegrass	Not available	Not available
GB59	FEID-AGSP	Idaho fescue-bluebunch wheatgrass	Not available	Not available
GB9111	POSA12-DAUN	Sandberg’s bluegrass-onespike oatgrass	Not available	Not available

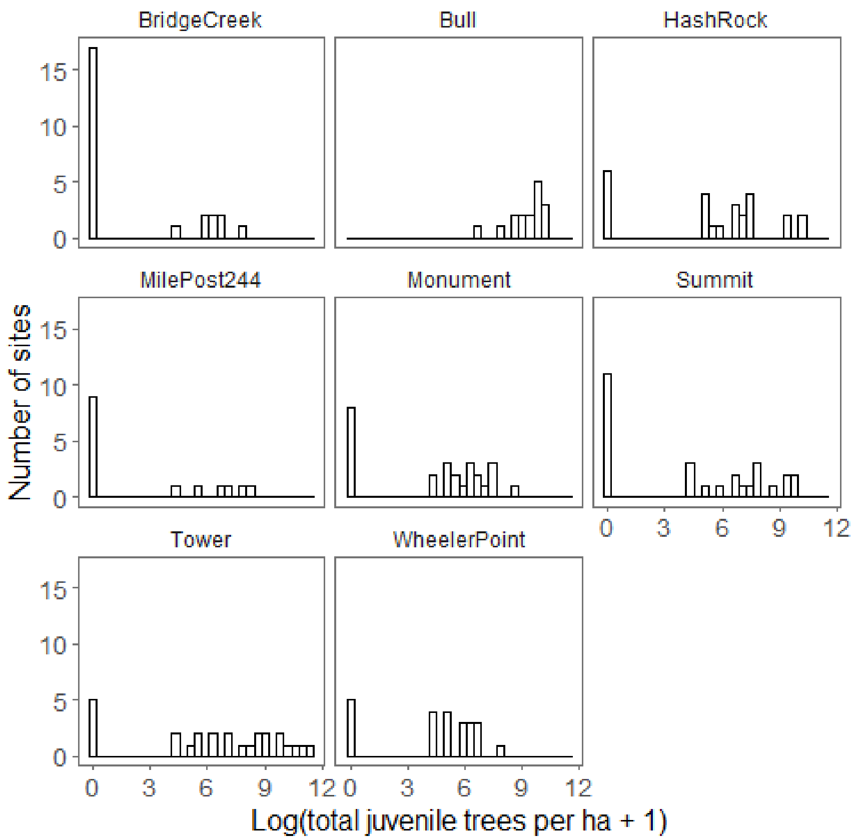

Figure B1. Distributions of site-level conifer densities across all fires. Note the abundance of zero counts.

Table B2. Proportion of sites in which each species of juvenile conifer were found.

Juvenile conifer species	Sites present (N = 184)
Ponderosa pine	84 (46%)
Douglas-fir	68 (37%)
grand fir/white fir	45 (24%)
western juniper	34 (18%)
western larch	30 (16%)
lodgepole pine	26 (14%)
Engelmann spruce	7 (4%)
subalpine fir	6 (3%)

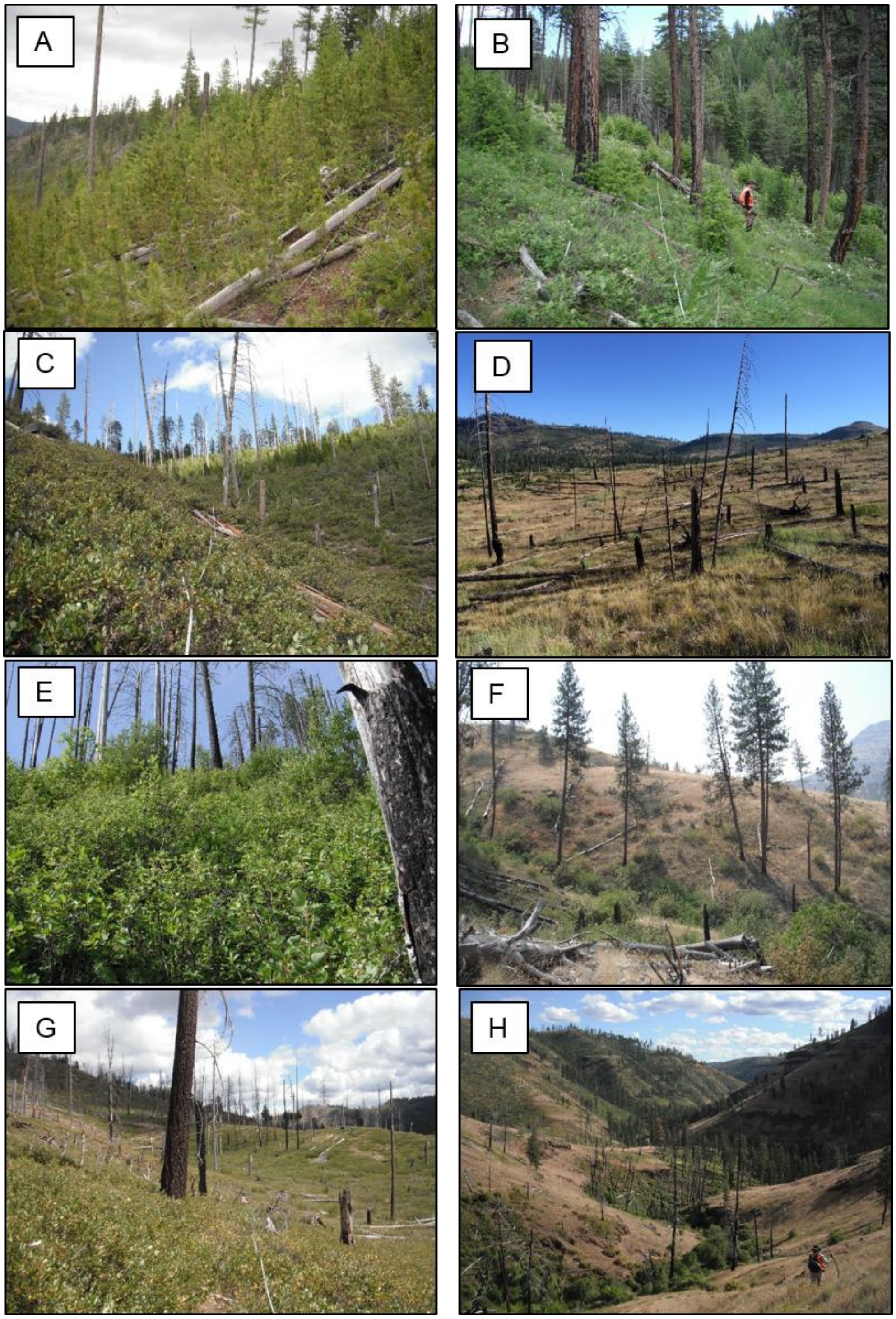

Figure B2. Illustrative photos of medium or high severity burn areas in each fire surveyed. While these are not typical of all sites in each fire, they illustrate the diversity of site types and vegetation responses: A) Bull: Example of abundant lodgepole juveniles; B) Summit: Bark scorch is visible on trunks of surviving overstory trees; C) Tower: Note abundance of fire-adapted shrub *Ceanothus velutinus*; D) Wheeler Point; E) Milepost 244: Note advanced, diverse shrub response; F) Monument Complex: Gentle topography and widely spaced trees contributed to a mosaic of low and medium burn severities; G) Hash Rock: Note abundance of fire-adapted shrub *Ceanothus velutinus*; H) Bridge Creek: Note topographic controls on vegetation growth.

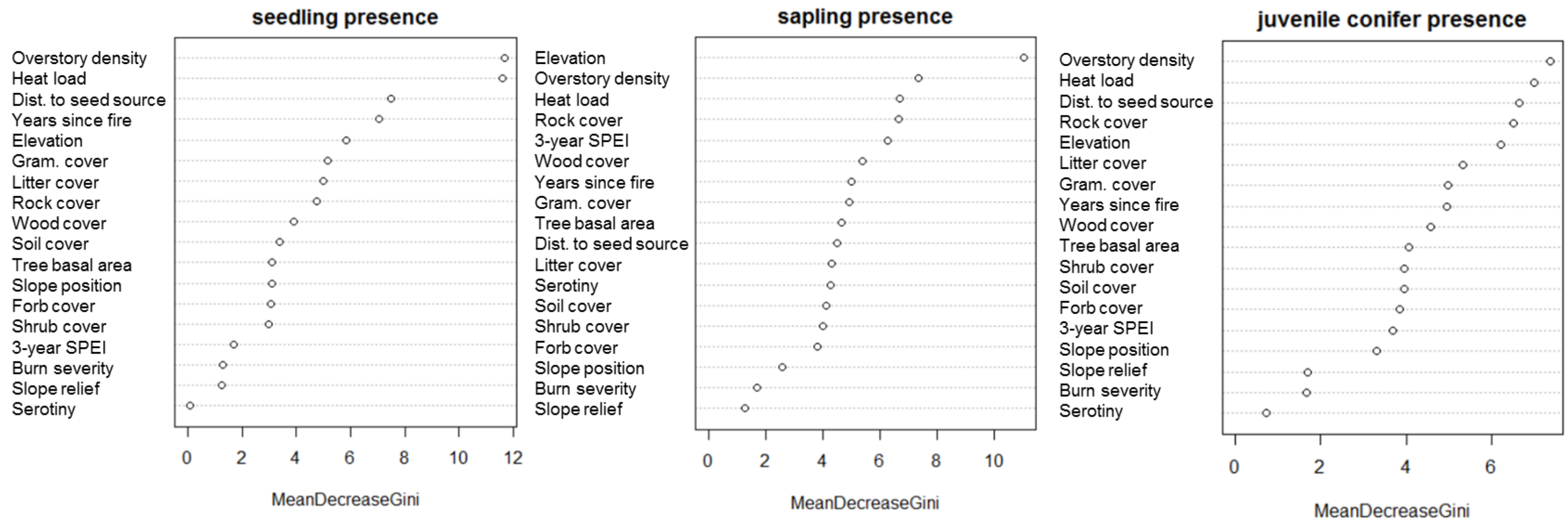

Figure B3. Variable importance plots from Random Forests analysis predicting presence of A) seedling; B) sapling; and C) all juvenile conifers. A higher mean decrease in the Gini coefficient indicates a variable is more important for classifying juvenile presence.

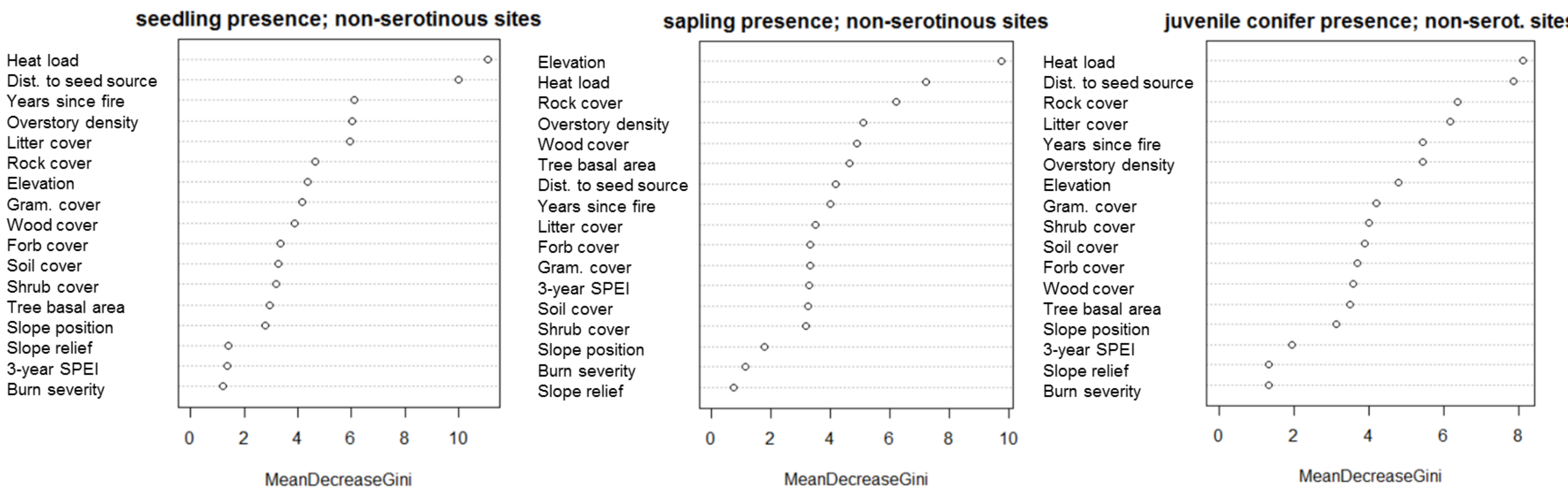

Figure B4. Variable importance plots from Random Forests analysis predicting presence of A) seedling; B) sapling; and C) all juvenile conifers, with serotinous sites omitted. A higher mean decrease in the Gini coefficient indicates a variable is more important for classifying juvenile presence.

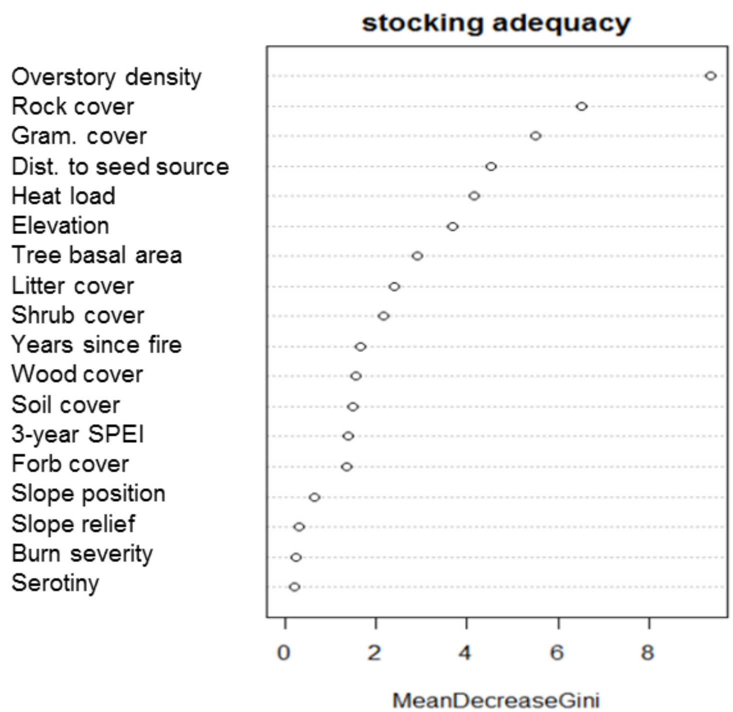

Figure B5. Variable importance plots from Random Forests analysis predicting whether a site meets the minimum stocking adequacy for a given forest type. A higher mean decrease in the Gini coefficient indicates a variable is more important for classifying juvenile presence.

Appendix C: Supporting Documents for Landowner Interviews

Interview Participant Recruitment Letter

Environmental Studies Program
1201 17th Street, 397 UCB
Boulder, CO 80309
University of Colorado, Boulder
Tel 720-212-6505 | http://www.colorado.edu/envs/

University of Colorado Boulder

Dear [Name],

You are invited to participate in a research project that is being conducted in cooperation with Oregon State University Extension, University of Colorado and University of New Hampshire. The main objective of the Communities and Forests in Oregon (CAFOR) study is to learn more about how landscapes in eastern Oregon are changing. We are interested in how land and forest management has changed on your land through the years, your perspective on the changes eastern Oregon has experienced, and how you plan to manage your land in the future.

As someone who owns forestland in eastern Oregon you have valuable insight that can contribute to this study. I am asking for your help in this research by sharing your experiences in an in-person interview. The interview will last as little as one hour to as long as three hours, depending on how much detail you wish to share or how complex your forest property is. The exact time and place of the interview can be arranged to fit your schedule, though it is preferable if the interview is conducted on your land.

The answers of all the study participants will be combined together into a research report that may help others better understand land management in the American West. **Your participation in this discussion is completely voluntary. You do not have to participate in this study if you do not want to.** Even if you agree to participate, you do not have to answer any question that you do not wish to answer or discuss. If you have any questions about this research, or about your rights as a participant, I will be happy to try to answer them. **All responses will be strictly confidential**.

Special precautions have been established to protect the confidentiality of your responses. After the interview, all of your identifying information will be removed. With your permission, I will tape record the discussion, because it is sometimes difficult to take accurate notes when speaking for an extended period of time. I will erase the tapes after I have finished taking the notes I need for the research. I will also ask you to make notes on a map of your land about any changes you have noticed on the landscape, as well as any management activities you have completed in the past or have planned for the future. I will make a digital copy of these notes and you will be able to keep the written copy for your records.

Information resulting from your interview will be identified by an identification code number, and not by your name or other personal information. However, other participants may repeat questions and responses outside of the interview setting. Only a small sample of individuals will be asked to participate in interviews, so your participation is important to this study. If you do not want to participate and do not wish to be contacted further, simply discard this mailing, and you will not receive any further correspondence. There are no foreseeable risks to you as a participant in this project, and you may find it useful to reflect on your forest and property management with an unbiased observer.

Your participation is extremely valuable. If you are willing to participate in an interview, or have questions regarding this study, please contact **Angela Boag** at **(720) 212-6505** or by email at **angela.boag@colorado.edu**. You may also contact the director of the Communities and Forests in Oregon (CAFOR) Project (www.cafor.weebly.com), Dr. Joel Hartter, at the University of Colorado at (541) 908-5334 or joel.hartter@colorado.edu.

Thank you for your help. I truly appreciate your cooperation and I hope to hear back from you.

Sincerely,

Angela Boag
Graduate Research Assistant
Environmental Studies Program
University of Colorado, Boulder
Cell: (720) 212-6505
Email: angela.boag@colorado.edu

Interview Guide

Respondent Information

- Interview ID
- Date and time
- Latitude/longitude
- Community, County
- Sex
- Have you participated previously in a research survey or interview?
- Years lived in this county
- Years owned this property
- Are you a full-year or seasonal resident?
- Property
    - Did you inherit or buy your property?
    - Total acres
    - Forested acres
    - Grazing acres
    - Agricultural acres
    - Own other property in Oregon?

Management Objectives, Plans and Actions

1.0 What are your primary land management objectives?

2.0 Do you have a forest management (or stewardship) plan?

If yes:

2.0.1 How far into the future does it plan?

2.0.2 Does it include a forest inventory?

2.0.3 Did you prepare it, or get help?

2.0.4 If helped, by whom?

2.1 Overall would you say you plan forest management actions far in advance (how far)?

3.0 Which of the following management actions have you taken in the last 10 years?

Precommercial thinning, commercial thinning, ladder fuel removal, ground fuel removal, prescribed burns, forest pest management, timber sale, planting seedlings, slash pile burning, mastication/chipping, other (ask to describe)

3.1 How large of an area did you take these actions on?

3.2 Do you do management work yourself or hire a contractor (or both)?

3.3 Do you know what your reentry time is?

3.4 What are the different forest types on your property?

3.5 What actions, if any, do you have planned on your land in the next 5 years?

If yes: 3.5.1 At what scale?

3.6 Do you actively manage to prevent or control noxious weeds?

4.0 Does the forest products market and economy overall affect your forestry activities?
If yes: 4.0.1 How?
4.1 Have public land management decisions affected you or your neighbors?
If yes: 4.1.1 How?
4.2 How well do you know the folks who own neighboring land?
4.3 Do you know if any of your neighbors have done/are doing active forest or grazing management?

Perceptions of environmental change and adaptation actions
5.0 What do you think makes a healthy forest?
5.1 Have you noticed any parts of your own forest or forest on neighboring land that looks unhealthy? If so, do you know why?
5.2 Do you take specific actions to reduce spread of pests and disease?
5.3 Are you concerned about any particular tree or shrub species on your land, either because of high mortality or because they're encroaching on certain areas?
5.4 Have you had any wildfires on your land or nearby?
If yes:
    5.4.1 When did the fire(s) occur?
    5.4.2 How large were they and how long did they burn?
5.5 Do you think your forests are at risk of wildfire? How great do you feel the risk of wildfire is on your property?
5.6 Do you take any specific actions to lower the risk of wildfire on your land?
5.7 Has the threat of wildfire changed in any way since you first moved here?
5.8 Do any of your management decisions involve planning for changes in wildfire threat in the future?
5.9 Are you worried about wildfire or insects/disease spreading from neighboring land to your property?

6.0 (If grazing): Have you experienced any challenges getting enough good water for the cattle?
If yes: 6.0.1 When did you have these difficulties, and what did you do?
6.1 (If grazing/agriculture): Have you experienced any problems with drought affecting your forage/crops?
If yes: 6.1.1 When did you have these difficulties, and what did you do?
6.2 (If not grazing/no agriculture): Have you had any problems associated with drought since you've lived here?
6.3 Are your reservoirs, wells or ponds different levels than in the past?
If yes: 6.3.1 When? What did you do?
6.4 Are you concerned about decreased water availability in the future?
If yes: 6.4.1 Why?
6.5 Are you at all concerned about the effects of your neighbors on your water supply or water quality?
If yes: 6.5.1 Why?

6.6 Have you had any problems with flooding since you lived here?
If yes: 6.5.1 When? What did you do?
6.7 Have you noticed more or less snow over the years, or snow melting at different times since you've lived here?
If yes: 6.7.1 Is this a concern for you?

7.0 Have you noticed any changes in temperature in this region since you've lived here?
If yes: 7.1.1 Is this a concern for you?

Beliefs and attitudes regarding climate change
8.0 Do you think that climate change is happening? Why or why not?
8.1 Do you think climate change is affecting temperatures, precipitation, drought, or insects and disease on your property? Overall in eastern Oregon?
If yes: 8.1.1 Do you think this will become a bigger management challenge in the future? Why or why not?

Engagement and Resource Needs
9.0 Have you participated in any activities with OSU Extension Service, stewardship organizations or forest collaboratives in the last 5 years, like workshops or meetings?
If yes: 9.0.1 How often do you participate?
9.1 If you had a question about your forest who would you call?
9.2 Do you get any regular newsletters from forest management or agriculture groups?
9.3 Do you need any additional resources to manage your land differently or better than the way you do now?
9.4 Are there any other issues you'd like to talk about?
9.5 Do you know anyone else who might be interested in participating in this research?

Demographic Information

- Age
- Current occupation
- Past occupation
- Highest education level attained
- Do you plan to leave this in the next 5 years?
- Do you know if you will eventually sell your land or pass it on to someone?
- In terms of your political views, do you consider yourself a Republican, Democrat, or Independent? If Republican, do you support the Tea Party?

CODEBOOK: Climate Adaptation and Non-industrial Private Forest Owners in Eastern Oregon

Codes corresponding to outcomes:

ACTNS – forest management actions taken in last 10 years

- PCT – Pre-commercial thinning
- CT – commercial thinning
- TIMSALE – timber sale
- GRNDR – ground fuel removal
- LADR – ladder fuel removal
- PLBURN – slash pile burning
- CHPNG – chipping slash and spreading/taking away
- INDTR – removal of individual dead/diseased/infested trees
- RXBRN – prescribed burn
- REFOR – reforestation: planting seedlings either to supplement natural regeneration or as restoration following wildfire/logging

EXTNT – extent of active management on property

- WHLPROP – whole property
- PROPSECT – large section of property
- STAND – stand-level active management
- INDTRL – individual tree-level management

PLNACTNS – planned future actions

- NOACT – no actions
- MAINT – maintain outcome of past treatments
- CONT – continue past treatment actions elsewhere on property/re-enter same location at a later date

FMP – have a forest management plan

- HQFMP – high quality FMP written within last 10 years with forest inventory
- LQFMP – low quality FMP, either outdated or partial (missing forest inventory)
- INFMP – informal FMP – either “in my head” or otherwise not formalized
- NOFMP – no forest management plan

PLNHORZN – planning horizon

- 30PLUS – plan more than 30 years into the future
- 20TO30 – plan twenty to thirty years into the future
- 10TO20 – plan ten to twenty years into the future
- 0TO10 – plan zero to ten years into the future
- OPP – take action spontaneously as opportunity arises

ADPTN

- IMPADPT – implemented climate change adaptation action
- SUGG – adaptation action suggestion/intention
- INTADPT – implemented/suggested adaptation is climate change-motivated (intentional)
- INCADPT – implemented/suggested adaptation is not climate change-motivated (incidental)

- CCIMPF – may consider adaptation options in the future when climate change impacts appear
- TIMBD – concerns about maintaining timber density while also keeping trees healthy
- NOCHG – would not change forest management practices regardless of climate change

Codes corresponding to subjective barriers to adaptation:

BELV – belief in anthropogenic climate change

- NOCC – there is no climate change occurring
- DKBEL – do not know what they believe/cannot say
- BELVDK – believe it is happening but do not know why
- NATCY – it is a natural cycle
- ANTHR – it is anthropogenic climate change
- ANTHNAT – it is a combination of natural cycles and anthropogenic climate change

PRCV – perceives/experiences local climate changes

- WRMG – perceives warming locally
- DRYG – perceives more frequent drought
- LOSNW – lower snowpack in recent years

FIRERSK – self-assessed risk of wildfire on own property

- NORSK – not at risk of wildfire
- LORSK – low risk of wildfire
- MEDRSK – medium risk of wildfire
- HIRSK – high risk of wildfire

INCRSK

- YINDRGHT – yes, wildfire risk is increasing because of drought
- YINSUPFAIL – yes, wildfire risk is increasing because of fire suppression failure
- YINC – yes, wildfire risk is increasing but I am not sure why
- NOINC – no, wildfire risk is not increasing

CCIMP – ways in which climate change will impact forests in eastern Oregon

- INFDEF – information deficit/uncertainty; climate change is happening but cannot predict what local impacts will be
- DRTSTRSS – will cause drought stress
- INCBUGS – anticipate more insect and disease issues
- INCFIRE – anticipate more large wildfires in future
- SPP – may change which species can live where

Codes corresponding to structural barriers to adaptation:

TIMBMRKT

- NOTIMR – timber markets do not impact management
- MILLSHT – Mill shutdowns have increased hauling costs or otherwise raised costs (e.g., no competition between mills so log prices are lower)
- HAULCSTS – haul costs impact management (e.g. high price of diesel, high pay for truckers)
- LOGPRIC – log prices too low
- MINVOL – forest owners has too little volume to sell to a mill
- NOMRKSMD – no market for small diameter timber

RSRCNEEDS – resource needs

- MONEY – need money for adaptive management (e.g. grants, cost-shares etc.)
- TIME – need more time to do management
- EQUIPMENT – need new/different equipment (e.g. to rent, hire or grants for buying)
- LABOR – need assistance performing actions
- EDUCATION – need more/different education on management strategies
- BETTERMRKTS – need better log markets to provide economic incentives for management
- TOPO – topography is too challenging to treat

NETWRK – organizations/networks individual engages with to gain resources

- OSUEXT – OSU Extension
- ODF – Oregon Department of Forestry
- PERSEXP – Personal experience (self or close family/friend)
- NRCS – Natural Resource Conservation Service
- OSWA – Oregon Soil and Water Association