Wildfire and Climate Change in Mixed-Conifer Ecosystems of the Northern Rockies: Implications for Forest Recovery and Management

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Authorization to Submit Dissertation

This dissertation of Kerry Kemp, submitted for the degree of Doctor of Philosophy with a Major in Natural Resources and titled "Wildfire and Climate Change in Mixed-Conifer Ecosystems of the Northern Rockies: Implications for Forest Recovery and Management," has been reviewed in final form. Permission, as indicated by the signatures and dates below, is now granted to submit final copies to the College of Graduate Studies for approval.

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Abstract

As disturbances continue to become more frequent and extensive with climate change, increasing concern is mounting about the ability of dry-mixed conifer forests to recover after wildfire. This concern stems in part from past management strategies, which have impacted the resilience of these forests. As such, future actions that managers propose to deal with climate change impacts will inevitably affect future resilience of these forests. My dissertation examined how climate, disturbance, and landscape variables influenced tree regeneration in dry mixed-conifer forests of the northern Rocky Mountains, using field data combined with downscaled climate data and satellite-derived burn severity data to characterize post-fire seedling regeneration across environmental gradients. Additionally, I examined how forest managers are thinking about climate change impacts and the adaptation measures they are considering to deal with these changes using a combination of breakout group discussions during workshops, interviews and surveys. Distance to a live seed source was one of the most important variables influencing the potential of post-fire regeneration after recent fires. The heterogeneity of the burned mosaic insures that most (> 80%) of the burned landscape is within a distance to live trees for successful regeneration, suggesting high resilience of these forests to recent fire. As climate continue to warm, however, temperature may outweigh the influence of seed source availability on seedling regeneration and the post-fire environment may no longer be favorable for regeneration in much (80%) of the existing dry mixed-conifer zone. Managers desire local climate change predictions that will help them identify thresholds for species resistance or resilience to propose effective management actions. These types of data will help managers move from using current management strategies to using more novel and appropriate techniques to help forests remain

resilient to a variety of uncertain future changes. Understanding the diverse and interacting ecological and social factors that influence the recovery or decline of dry mixed-conifer forests will increasingly improve predictions about the future impacts of disturbance, climate change, and management.

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Dedication

To my parents, Phil and Mary Kemp, for sharing your love of nature with me and for teaching me to be curious, think analytically, and persevere through challenges. Your dedication to my education and extracurricular enrichment throughout my childhood is a testament to my current success.

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CHAPTER 1

Dissertation Overview

Climate change and wildfire: Interactions and complexities

Globally, forests are becoming increasingly vulnerable to climate change (Allen et al. 2010, Allen et al. 2015, Millar and Stephenson 2015). Increasing temperatures have led to more intense droughts (Breshears et al. 2005, Allen et al. 2015), extensive pathogen and insect outbreaks (Weed et al. 2013), and more frequent large wildfires (Westerling et al. 2006, Liu et al. 2010, Jolly et al. 2015). While forests have evolved to persist in the face of disturbance, recent intensification of disturbance processes has led scientists to question how much disturbance can be tolerated before forest resilience is undermined (Millar and Stephenson 2015, Trumbore et al. 2015). As novel climate conditions continue to shape, change, and interact with forest disturbances, forest managers are increasingly being tasked with predicting impacts to forest ecosystems at local levels while also being asked to integrate adaptation and mitigation measures into planning efforts to deal with these changes (Millar et al. 2007). Yet, land managers and scientists lack an integrated understanding of how climate and disturbance will interact to influence future forest structure and composition.

Wildfire is one of the most significant disturbance processes structuring temperate forests in the western United States (e.g., Romme 1982, Agee 1993, Romme et al. 1998). Wildfire can influence heterogeneity in vegetation across a broad regions (Lentile et al. 2005) by modifying succession and recovery through time (Turner et al. 1993, Turner and Romme 1994). Fire regimes, or the attributes and characteristics defining wildfire within a particular landscape or vegetation type over a given time period, characterize the frequency, extent, severity and intensity of a given fire event (Agee 1993, Baker 2009). Feedbacks among wildfire, climate, and vegetation at various spatial and temporal scales shape how fire regimes and vegetation respond to changes in climate (e.g., Whitlock et al. 2003, Higuera et al. 2009). For example, at centennial to millennial times scales, shifts in vegetation composition in response to climate may drive changes in fire regimes (Higuera et al. 2009); at shorter time scales, shifts in fire frequency, extent, or severity in response to climate may influence the composition of regenerating vegetation (Johnstone et al. 2010). The resilience of forest ecosystems to changes in climate and fire may depend upon how climate-fire interactions are manifested at local and regional scales (Millar and Stephenson 2015, Trumbore et al. 2015).

Ecosystem resilience emphasizes the ability of an ecosystem to absorb disturbance while maintaining qualitatively similar structure and function (Holling 1973, Groffman et al. 2006, Moritz et al. 2011). Ecosystem resilience is often conceptualized using a ball-and-cup model in which an ecosystem (the “ball”) oscillates within certain conditions until a particular event or disturbance induces the ball to shift its position either beyond the bounds of its original conditions (or “cup”) into a qualitatively different state or back to its initial state (Figure 1.1; Gunderson 2000). For example, a forest that experiences a disturbance within the range of conditions for which it is naturally adapted will be able to recover its structure and function shortly following the disturbance (Trumbore et al. 2015). In contrast, forests that are degraded or experience a disturbance of unusually high severity, frequency, or extent may experience delayed recovery or shift into a new vegetation state (e.g., Odion et al. 2010). The point of transition between the two states is often referred to as a threshold. Thresholds may be increasingly important to define and characterize in order to understand

and manage for forest ecosystem response to climate change (e.g., Groffman et al. 2006).

Drivers of change in dry mixed-conifer forests

Dry-mixed conifer forests in the western United States are increasingly experiencing multiple stressors that threaten to undermine their resilience. Over the past several decades extensive wildfires have contributed to millions of hectares of forests burned every year (Stephens et al. 2014). Although dry forests evolved with frequent fire (e.g., Swetnam and Baisan 1996, Taylor and Skinner 1998, Heyerdahl et al. 2008b, Falk et al. 2011), concern is mounting about potential uncharacteristically severe burned areas within these large wildfires as a result of fuel buildup from prior management activities (e.g., Covington 2000, Stephens et al. 2013, Williams 2013). Warming climate conditions will exacerbate conditions for extreme wildfires (Liu et al. 2010, Westerling et al. 2011, Moritz et al. 2012) while also influencing habitat suitability for mature and regenerating trees (Bell et al. 2014, Dobrowski et al. 2015). As a result, restoring the resilience of dry-mixed conifer forests has become a high priority for forest managers, especially given the potential consequences of climate change (Millar et al. 2007, Stephens et al. 2013).

Extensive area burned in recent decades in forests of the U.S. northern Rocky Mountains (referred to as northern Rockies from here forward) may reflect the combined effects of changing fuel structure and availability as well as climate (Higuera et al. 2015). Past logging, grazing, and fire suppression activities (Hessburg and Agee 2003) fostered tree regeneration, increased tree density and shifted species composition towards more shade-tolerant species (Hessburg et al. 2000, Keeling et al. 2006, Naficy et al. 2010). Low annual area burned in the northern Rockies in the mid-20th century also coincides with less conducive climate conditions for burning (Morgan et al. 2008, Higuera et al. 2015).

However, in recent decades, warmer springs and earlier snowmelt have increased fire season length and fire occurrence, particularly in the northern Rockies (Westerling et al. 2006, Klos et al. *2015*). In the past 60 years, regional mean annual temperatures have increased 1°C (Mote and Salathe 2010).. Climate projections predict an additional increase of 2-5°C by 2090, with disproportionate warming and drying during the summer (Mote and Salathe 2010). Because widespread fires in this region occur in years with warm springs and high summer temperatures associated with summer drought (Morgan et al. 2008), projections of increased temperatures, drought, and fire danger (Jolly et al. 2015) suggest that, in combination with changes in fuels, lower elevation dry forests may become increasingly vulnerable to more frequent and extensive fires (Littell et al. 2010).

Changes in the scale and distribution of fire effects may be key to determining whether recent large wildfires will compromise the resilience of forest ecosystems (Stephens et al. 2013). Burn severity, or the degree of ecological change as a result of fire (Lentile et al. 2006), is likely to be one fire effect that impacts resilience. In the northern Rockies, dry mixed-conifer forests historically burned in low- and mixed-severity fires (Arno et al. 2000, Baker et al. 2007, Heyerdahl et al. 2008a, Heyerdahl et al. 2008c), however, high-severity effects were likely present at some scale in all of these fires (Pierce et al. 2004, Stephens et al. 2013, Odion et al. 2014). Although many of the tree species in these ecosystems have evolved with and adapted to withstand low and moderate severity fire, the size of high-severity patches or return interval of frequent fires may limit tree reestablishment by limiting seed sources across large areas (Stephens et al. 2013, Millar and Stephenson 2015). The capacity of forests to recover from these fires will depend on traits and adaptations of individual species for resistance or recovery (Baker 2009). To the extent that recent and future fires depart from historical conditions that each species is adapted to, the potential for a shift in vegetation or delaying forest recovery will increase.

Goals and objectives

In my dissertation, I examined interactions among climate, fire, and post-fire tree regeneration in dry mixed-conifer forests and social perceptions shaping forest management in the northern Rockies. I aimed to identify mechanisms and thresholds that contribute to the persistence and resilience of those forests in the face of changing climate and fire regimes. As forests continue to experience future changes in disturbance and climate, forest managers must anticipate how climate change impacts the resources they manage. Decisions that managers make could have the potential to influence the trajectory of these forests. Therefore, I also examined how forest managers are thinking about and managing for the impacts of climate change.

In Chapter 2, I characterized the response of post-fire tree regeneration to fire legacies (e.g., burn severity, distance to seed source), topographic position (e.g., elevation, heat load index), and biotic environmental variables (e.g., tree canopy cover, tree density). I sampled tree seedling regeneration in 21 different large fire events across gradients in burn severity, elevation, aspect and latitude to understand: (1) How abiotic landscape conditions and fire characteristics determined the presence and density of post-fire tree seedlings; (2) What factors limited species-specific regeneration post-fire and whether different factors influenced the presence and abundance of post-fire regeneration; and (3) What proportion of the landscape exhibited structural characteristics resilient to large mixed-severity fires?

In Chapter 3, I aimed to understand how fire and climate interact to influence suitable habitat for post-burn tree seedling regeneration currently and in the future. These interactions may be key for identifying thresholds of resistance and resilience to future change. I used post-fire seedling density data collected from a wide variety of sites characterizing dry mixed-conifer forests in the northern Rockies to understand how seedling abundance varied in response to climate and burn severity. Specifically, I identified sampled locations across the dry mixed-conifer forest ecotype that may fail to regenerate after future fires and experience a shift in vegetation change in response to disturbance and climate change.

Managing forests will become increasingly challenging as climate change continues to make forest response to disturbance and environmental stressors uncertain. In Chapter 4, I focused on how managers in the northern Rockies are using climate change science in their planning efforts and thinking about the impacts of climate change on forest resources. As part of an interdisciplinary team, I helped conduct interviews, surveys, and focused workshops on climate change impacts to forests with forest managers throughout the northern Rockies. Using this mixed-methods approach, I examined (1) How climate-change science is useful for U.S. Forest Service and Bureau of Land Management resource managers' work and whether they as individuals, or their agencies, are currently incorporating this information into land management planning; (2) What management actions resource specialists see as effective for adapting to, or mitigating, climate change and if their agencies are considering implementation of these actions; and (3) What barriers resource managers' perceive as impeding their use and incorporation of climate science into management.

Finally, in Chapter 5, I summarized the implications of this work for forest managers trying to ensure the resilience and regeneration of forests in the face of climate change and after fire. I also point to future research needs that would help scientists and managers continue to understand how processes such as changing climate and disturbances impact patterns of forest mortality and regeneration. I identify thresholds of patch size and temperature that impact forest resilience and provide quantitative information that will help the scientific community predict how feedbacks among climate, disturbance, and topography may influence future tree species contractions. .

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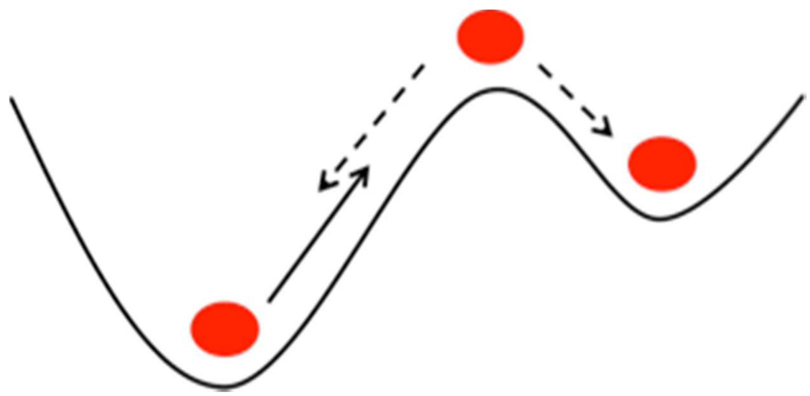

Figure 1.1. Conceptual diagram of the "ball-in-cup" model, where each basin represents a qualitatively different stable state. Disturbance can induce the ball to shift its position, from which it can either return to its initial state, or into a new basin (or qualitatively different state). The point of this transition is often termed an ecological threshold (Adapted from Gunderson 2000, Groffman et al. 2006).

CHAPTER 2

Fire Legacies Impact Conifer Regeneration Across Environmental Gradients in the U.S. Northern Rockies

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Abstract

An increase in the incidence of large wildfires worldwide has prompted concerns about the resilience of forest ecosystems, particularly in the western U.S., where recent changes are linked with climate warming and 20th-century land management practices. Objectives. To study forest resilience to recent wildfires, we examined relationships between fire legacies, landscape features, ecological conditions, and patterns of post-fire conifer regeneration. We quantified regeneration across 182 sites in 21 recent large fires in dry-mixed conifer forests of the U.S. northern Rockies. We used logistic and negative binomial regression to predict the probability of establishment and abundance of conifers 5 to 13 years post-fire. Seedling densities varied widely across all sites (0 – 127,500 seedlings ha^{-1}) and were best explained by variability in distance to live seed sources ($\beta = -0.014$, $p = 0.002$) and pre-fire tree basal area ($\beta = 0.072$, $p = 0.008$). Beyond 95 m from the nearest live seed

source, the probability of seedling establishment was low. Across all the fires we studied, 75% of the burned area with high tree mortality was within this 95-m threshold, suggesting the presence of live seed trees to facilitate natural regeneration. Combined with the mix of species present within the burn mosaic, dry mixed-conifer forests will be resilient to large fires across our study region, provided that seedlings survive, fires return intervals do not become more frequent, high-severity patches do not get significantly larger, and post-fire climate conditions remain suitable for seedling establishment and survival into the future.

Keywords Tree regeneration • Mixed-severity • Wildfire • Patch size • Distance to seed source • Resilience

Introduction

Large wildfires have been increasing worldwide over the past several decades (Kasischke and Turetsky 2006; Westerling et al. 2006; Pausas and Fernández-Muñoz 2012), a pattern predicted to continue in many forested regions with even moderate climate warming (Flannigan et al. 2009; Pechony and Shindell 2010; Littell 2011; Rogers et al. 2011). Large disturbances, including wildfires, shape ecosystem structure and function for decades to centuries, and shifts in their frequency, size, or intensity can have unknown implications for forest resilience (Turner 2010). We use the term "resilience" to describe the capacity of a system to absorb disturbance without transitioning into a qualitatively different state, emphasizing the maintenance of system structure and function (Holling 1973; Groffman et al. 2006; Moritz et al. 2011). Large, intense wildfires can reduce forest resilience by shifting post-fire species assemblages (Johnstone et al. 2010a,b) or initiating type conversions to non-forest vegetation (Savage and Mast 2005; Odion et al. 2010). Thus, understanding forest resilience to future wildfires, particularly in the context of climate change, depends upon identifying the mechanisms that influence tree regeneration, survival and composition.

Dry mix-conifer forests of the western U.S. may be particularly vulnerable to ongoing and future shifts in wildfire activity (e.g., Williams et al. 2010), given the combined effects of 20^{th}-century land use and land management practices on species composition, fuel loads, and fire regimes (e.g., Hessburg et al. 2000; Keeling et al. 2006; Naficy et al. 2010). Of particular concern in these ecosystems is the possibility that large, stand-replacing wildfires will remove viable seed sources over large areas, significantly delaying or preventing post-fire forest recovery (Stephens et al. 2013) and converting forested areas into a qualitatively different vegetation type. However, given that nearly all wildfires include patches of stand-replacing fire interspersed with low- to- moderate severity patches (Turner and Romme 1994; Baker et al. 2007; Odion et al. 2014), an alternative scenario is that spatial heterogeneity in fire effects will allow for forest recovery and resilience to fire (Halofsky et al. 2011).

Forest recovery, or conversely, type-conversion, after large mixed-severity wildfires depends upon a combination of factors, including spatial variability in fire effects, pre-fire species composition, species-specific persistence mechanisms, and post-fire abiotic (e.g., topography, climate) and biotic (e.g., canopy opening, competition) conditions (Stephens et al. 2013). In forests where fire-resilient and fire-resistant (Baker 2009; Keeley et al. 2011) species co-occur, like dry mixed-conifer forests in the U.S. northern Rockies, the diversity in species and stand structure may increase forest resilience to variable fire effects (Halofsky et al. 2011). Fire-resistant species, such as Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), have thick bark and high crown base heights that facilitate survival of low-intensity surface fires and successful regeneration into patches burned at low and moderate severity where many large trees survive. Resprouting species, such as quaking aspen (*Populus tremuloides* Michx.), or serotinous species, such as lodgepole pine (*Pinus contorta* Douglas ex Loudon), however, are more likely to regenerate after stand-replacing fire (Turner et al. 1997; Franklin and Bergman 2011; McKenzie and Tinker 2012). Understanding how different landscape patterns of mixed-severity fires impact forest structure and regeneration can be particularly challenging, however, in part because of high variability in pre- and post-fire conditions and variability within and among fire events.

Studies of conifer regeneration from single mixed-severity fire events (e.g., Lentile et al. 2005; Donato et al. 2009; Crotteau et al. 2013) or small geographic regions (e.g., Shatford et al. 2007; Collins and Roller 2013), have helped highlight the singular importance of burn severity, patch size, or abiotic conditions to post-fire regeneration. It remains unclear, however, how these results scale up to multiple fire events across broad regions. Here, we quantified natural post-fire tree seedling regeneration in 21 large (> 400 ha), individual mixed-severity fire events across a 21,000 km^2 region, spanning most of the range of dry mixed-conifer forests in the U.S. northern Rockies. We combined field data with statistical modeling to quantify the relationships among post-fire seedling abundance and composition, wildfire patch metrics, and abiotic and biotic variables. The broad extent of our study gives us the unique opportunity to identify important mechanisms that drive patterns of forest recovery at broad scales and infer the resilience of dry mixed-conifer forests to future large fires and climate change.

Methods

Study region

Our study region encompasses the range of dry mixed-conifer forests in the U.S. northern Rockies, spanning a four-degree south to north latitudinal gradient (Figure 2.1). Dry-mixed conifer forests in this region are dominated by Douglas-fir and varying proportions of ponderosa pine, grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), and lodgepole pine. Across the study region, average temperatures range from 13.8°C to 21.1°C in July and -6.8°C to -0.6°C in January, and total precipitation ranges from 398 mm to 886 mm (PRISM 2014). The study region experiences warm, dry summers and wet winters. Soils are dominantly inceptisols and entisols derived from granitic Idaho batholith parent material (USDA 2014).

Terrain is characterized by steep topography, with dramatic ecotone boundaries and steep elevation gradients that encompass multiple biomes from river valleys to ridgetops. South-facing hillslopes at low elevations are dominated by ponderosa pine with some Douglas-fir, while north-facing slopes at the same elevation can maintain a mix of ponderosa pine, Douglas-fir, grand fir, and lodgepole pine. Low elevation dry-mixed conifer forests dominated by ponderosa pine historically experienced surface fires with return intervals of years to a few decades (Heyerdahl et al. 2008a,b), while higher elevations and more mesic sites experienced less frequent, mixed- or even high-severity fires (Arno et al. 2000; Baker et al. 2007; Odion et al. 2014).

Sampling design

In the summer of 2012 and 2013, we sampled a total of 182 sites, stratified across the range of biotic and abiotic gradients characterizing dry mixed-conifer forests of the region.

To identify potential sampling sites, we used a geographic information system (ArcGIS 10.0) to randomly select points within defined elevation, aspect, and burn severity classifications, in large (> 400 ha) fires that burned in 2000 and 2007 (Gibson et al. 2014, Morgan et al. 2014). Thirty-meter resolution vegetation data from the LANDFIRE (2010) database and 10-m resolution digital elevations models (DEMs) were used to characterize forested regions within Idaho and western Montana. We used Relative differenced Normalized Burn Ratios (RdNBR; Miller and Thode 2007) derived from 30-m Landsat TM+ satellite imagery from the Monitoring Trends in Burn Severity (MTBS 2011) project to initially classify burn severity into four categories: (1) unburned-unchanged, (2) low severity, (3) moderate severity, and (4) high severity. Sites were classified as "unburned" if they did not burn in either 2000, 2007, or in the 28- or 29- years covered by the MTBS data (i.e., since 1984). Within all four burn severity classes, we selected potential sites across three equally distributed elevation bands, with northeast- or southwest-facing aspects. Additional criteria for potential sites included a minimum polygon size of 0.81 ha (3 x 3 30-m pixels) to account for imprecision in satellite-derived data, separation of at least 120 m from other sites to minimize the potential for spatial autocorrelation, and proximity (within 2.5 km) to roads or wilderness research stations to maintain accessibility. Fifteen percent of sites were within federally designated wilderness areas. In all cases, sites that had been salvage logged or planted post fire were excluded from sampling based on communication with local USDA Forest Service personnel. Our stratification resulted in sample sites which spanned the full range of climates characterizing dry mixed-conifer forests in the region, as represented by the ratio of actual to potential evapotranspiration (Appendix B; Figure 2.1).

At each site, burn severity was field verified using environmental evidence including estimates of percent tree mortality, bole scorch, and shrub stem mortality. If a site did not fall within the desired burn severity classification, or was inaccessible, the location was offset by 30 m in cardinal directions until the desired stratification was attained. Live tree seedlings were counted within a 60-m long belt transect of variable width of 1 to 10 m, with transect width determined prior to sampling each plot based on visual estimates of seedling density. Vegetation cover, overstory tree basal area, and tree canopy cover were measured at 0, 30, and 60 m along each transect and averaged for a site. Vegetation cover was classified by lifeform (shrub, forb, graminoid, tree) within 1 m^2 sub-quadrats. Overstory tree basal area was quantified using a 2 or 4 m^2 ha^{-1} basal area factor prism on a variable radius plot, and canopy cover was recorded using a spherical crown densiometer (Forestry Suppliers, Inc., No. 43887). Overstory tree species, diameter at 1.37 m height, and percent mortality were also recorded and used to calculate the density of live and dead trees for each species. We quantified the distance to a live seed source by measuring the horizontal distance to the ten nearest live seed trees of all species from the transect center using a laser range finder (Truepulse® 360 B/Laser Technology) and averaging these distances for a site. Seed trees were confirmed to either have cones or be large enough to be reproductively mature. Distances greater than 500 m could not be measured. Additional site level data included slope, aspect, latitude, longitude, and elevation. Slope, aspect, and latitude were used to estimate potential heat load from direct solar radiation by calculating a heat load index (following McCune and Keon 2002).

Statistical analysis

We used a two-stage modeling approach to examine patterns of natural seedling regeneration as a function of three categories of response variables: legacies of the fire (e.g., burn severity, distance to live seed source), abiotic environmental variables (e.g., elevation, heat load), and biotic environmental variables (e.g., vegetation cover, tree basal area, and tree canopy cover; Table 2.1). We first used a logistic regression model to predict tree seedling presence or absence and then used a negative binomial or zero-inflated negative binomial model to predict tree seedling abundance (count). Logistic and count models were compared to evaluate whether the processes influencing seedling presence and abundance differed. All analysis were completed in R version 3.0.2 (R Core Team 2013) using the "MASS" package (Venables and Ripley 2002) for the negative binomial models and the "pscl" package (Zeileis et al. 2008; Jackman 2012) for the zero-inflated models. The logistic regression models are part of the standard statistical package in R.

Logistic regression models for seedling presence-absence

Five logistic regression models were developed using a binomial distribution to predict the presence or absence of tree seedlings. These models included a model for all species present on the site, and four species-specific models for species present on > 15% of the sites: Douglas-fir, ponderosa pine, lodgepole pine, and grand fir. All predictor variables were tested for collinearity using Spearman's rank correlation. Tree canopy cover was dropped from each model because it was well correlated ($\rho > 0.5$) with overstory tree basal area and the average distance to a live seed source. Models were then constructed using the remaining predictor variables and a censor variable indicating whether seed sources were present and measured on a site (Table 2.2). We examined the sensitivity of the parameter estimates (β) using forward-backward stepwise selection with AIC selection criteria and determined there was little change in these estimates due to model reduction. Therefore, we kept the fully specified models with all variables for comparative purposes.

We assessed logistic model fit using the three summary measures: the deviance residual, a Hosmer-Lemeshow test, and the area under the curve (AUC) of a receiver operating characteristic (ROC) curve (Appendix B). The AUC for each model (Fawcett 2006) can vary from 0.5 (random) to 1.0 (perfect prediction), where 0.7-0.8 is “acceptable”, 0.8-0.9 is “excellent”, and 0.9-1 is “outstanding” discrimination between the model predictions and the observed data (Hosmer et al. 2013). Additionally, to discriminate between presences and absences based on the modeled probabilities, we used a classifier value calculated from the ROC curve that maximized the rate of true positives while minimizing false positives for each model (Fawcett 2006). Using this classifier value, we calculated the positive predictive rate (PPR; i.e., the ratio of true positives to the sum of true and false positives) and the negative predictive rate (NPR; i.e., the ratio of true negatives to total negatives) to evaluate how well models predicted the actual presence or absence of tree seedlings across our study sites. To avoid model overfitting, we subsequently used cross-validation techniques to evaluate the Hosmer-Lemeshow statistic and AUC for each model (Appendix B; Hosmer et al. 2013).

Count models for regeneration abundance

Our tree seedling count data had a high proportion of zeros (26% for all species combined; 35%, 68%, 74%, and 86% for Douglas-fir, ponderosa pine, lodgepole pine, and grand fir, respectively) and the distribution of seedling counts was strongly left-skewed. Therefore, we considered a number of alternative generalized linear models for discrete skewed data, following the procedures outlined by Zeileis et al. (2008). Because our data displayed significant overdispersion (e.g., the variance was larger than the mean), we modeled the abundance of all species and the abundance of Douglas-fir using a generalized linear model with a negative binomial distribution. We developed zero-inflated negative binomial models for the abundance of ponderosa pine, lodgepole pine, and grand fir (Appendix B; Martin et al. 2005; Zuur et al. 2009; Hilbe 2011).

To compare the variables influencing presence and abundance of tree seedling regeneration, we kept all of the predictor variables used in the logistic regression model in the count models. Zero-inflated models were parameterized with an intercept-only predictor function for the zero portion of the model (Appendix B; Zuur et al. 2012). In addition, an offset variable was included in each count model to correct for the variable sampling area. Five sites were removed from this analysis because we failed to record the transect area (n = 177).

We compared predicted and observed values from each of the count models using Spearman's rank correlation. Model fit was assessed visually by plotting the Pearson's residuals against the fitted values (Zuur et al. 2009). Furthermore, we performed a goodness-of-fit test using Monte Carlo simulations (n = 1000) to calculate the Pearson's chi-squared statistic, where the expected probabilities from the parameterized model are used to generate new observations set as the "observed" counts. These observed counts are then compared to the "expected" counts drawn from a random negative binomial distribution. A significant lack of fit is indicated by a p-value ≤ 0.05. We also calculated the Pearson's chi-squared test statistic and p-value using cross-validation for each model (n = 1000).

Patch size analysis

To determine the proportion of the landscape burned at high severity (i.e., stand-replacing wildfire), we used the classified MTBS data for the 21 sampled fire events and calculated the mean distance from each pixel classified as high severity to the nearest pixel of lower severity (i.e., classified as moderate, low, or unburned). Distance calculations were made from the raster images of the individual fires within R using the “gdistance” (Etten 2014) and “rgal” (Bivand et al. 2014) packages. Because pixel size in the MTBS dataset is 30 m, the minimum distance from a high severity pixel to an edge was 30 m if the two pixels were adjacent. We merged all fires for a single year (2000 or 2007) into a single raster and calculated the cumulative proportion of high burn severity pixels that were various distances from a pixel of lower burn severity. These distributions were compared to the average distance from live seed source trees measured on the ground at each of our high burn severity sites (n = 61).

Results

We counted over 10,000 seedlings of eight different species on 182 sites across our study region. Douglas-fir was the most abundant species, present on 120 sites (66%; Table 2.2). Total seedling densities ranged from 0 to 1.3×10^6 seedlings ha^{-1} and ranged four orders of magnitude for the most abundant individual tree species, from 0 to 3763 seedlings ha^{-1} (Figure 2.2). Seedling densities did not vary significantly between the two fire years (Mann-Whitney U test: W = 4378, p = 0.456). Nine sites had no seed sources within 500 m (the maximum detectable distance), and of these nine sites, only one had seedlings present, all of which were lodgepole pine (Figure 2.2). Conifer species composition did not vary significantly between trees present before fire and seedlings present after the fire, with the exception of a slight increase in lodgepole pine present in sites that burned in 2007, and a decrease in subalpine fir on sites where it was present and burned in 2007 (Table 2.2).

Models for conifer seedling presence

All species model

Distance to a live seed source was the most important variable predicting seedling presence (β = -0.014, p = 0.002; Table 2.3), with a lower probability of presence with increasing distance (Figure 2.3a). We identified an optimal classifier probability of 0.87 with the ROC analysis (Table 2.3), above which seedlings were predicted to be present. This probability threshold corresponded to a maximum distance of 95 m from a live seed source (Figure 2.3a). Sites with a higher pre-fire tree basal area also had a significantly higher probability of seedling presence (β = 0.072, p = 0.008; Table 2.3), though this pattern was most apparent at sites with low basal areas. All sites with a pre-fire tree basal area of 5 m^2 ha^{-1} had a high probability of seedling establishment, and above a basal area of 20 m^2 ha^{-1} this probability was > 95%.

Our model correctly predicted the proportion of sites with seedling presence 93% of the time (PPR) and seedling absences (NPR) 39% of the time. The model distribution fit the data well ($p > 0.08$; Table 2.4). An AUC value of 0.826 indicated an excellent ability to discriminate between sites with and without seedlings across our extensive sample region (Table 2.4), which was robust to cross-validation (Appendix C).

Species-specific models

Distance to a live seed source was the most significant variable influencing the presence of Douglas-fir and ponderosa pine seedlings (Table 2.3). Seed source distance was marginally significant for grand-fir, and insignificant for lodgepole pine (Table 2.3). The further a site was from a live seed source, the lower the probability of Douglas-fir, ponderosa pine, or grand fir presence (Figure 2.3b). Ponderosa pine seedlings were most likely to be present within 60 m of a live seed tree (Figure 2.3b). Similarly, Douglas-fir was most likely to be present within roughly 75 m of a live seed source. Grand fir presence was probable as far as 165 m from a live seed tree (Figure 2.3b). The censored variable, indicating whether or not a live seed source was measured on the site, was significant in these models only for ponderosa pine, lodgepole pine, and grand fir, with the odds of having seedlings present on sites without seed sources between three and seven percent less than sites with seed sources. Furthermore, distance to a live seed source was the only variable that was important in predicting the presence of ponderosa pine and grand fir tree seedlings across our study region (Table 2.3).

Elevation and heat load index were also important in the species-specific models for Douglas-fir and lodgepole pine (Table 2.3). Douglas-fir and lodgepole pine tree seedlings were more likely to be present at higher elevations and on sites with a low heat load index (Table 2.3). Tree basal area was important in predicting the presence of Douglas-fir on a site but did not influence the presence of any of the other species (Table 2.3).

Each of the full and cross-validated species-specific regression models predicting tree seedling presence performed better than random, as indicated by median AUC values greater than 0.5 (Table 2.4; Appendix C). Models correctly predicted the proportion of sites with seedling presence between 69 – 91% of the time (PPR), and correctly predicted the proportion of sites with seedlings absent (NPR) between 47 and 91% of the time (Table 2.4). Deviance chi-squared and Hosmer-Lemeshow statistics validated the fit of our data to a binomial distribution ($p > 0.05$), and were confirmed by cross-validation for most models (Appendix C).

Models for conifer seedling abundance

All species model

Distance to the nearest live seed source (β = -0.007; p << 0.001) and tree basal area (β = 0.032; p = 0.016) remained the most important predictors of seedling abundance in the negative binomial model (Table 2.3), with the same directionality whether predicting seedling presence or seedling abundance. There was significant fit between the data for our full all-species model and the negative binomial model distribution (Table 2.4), though this model cross validated poorly (Appendix C). Some variability between the predicted estimates and the observed seedling counts was present in the model (Table 2.4); the negative binomial all-species model tended to overpredict the abundance of seedlings on sites. Forty-five percent of the observed seedling counts fell within the range of predicted counts, +/- two standard errors (Appendix C; Figure C1).

Species-specific models

While the variables determining abundance of tree seedlings in the all species model and the ponderosa pine model did not change from the corresponding models predicting seedling presence, additional variables helped predict the abundance of Douglas-fir and grand fir across the study region (Table 2.3). For example, Douglas-fir abundance was significantly higher on sites with a longer time since fire (Table 2.3). Burn severity was also significant for Douglas-fir abundance; the expected counts of Douglas-fir seedlings were 8 and 13 times higher in sites burned at low and moderate severity, respectively, than in unburned sites. Although elevation and heat load influenced lodgepole pine presence, these variables were not important for determining its abundance (Table 2.3). Grand fir abundance was negatively related to both vegetation cover and tree basal area and distance to a live seed source was not important for its abundance on a site (Table 2.3).

The regeneration count models performed moderately well. Correlation between the predicted and observed counts ranged from 0.4 to 0.7 (Table 2.4). The negative binomial models correctly predicted 41 to 50% and overpredicted between 31 and 60% of observations for each species. A negative binomial distribution and a zero-inflated negative binomial distribution fit the observed data for all the full species-specific models except ponderosa pine, as indicated by the Pearson's chi-squared goodness-of-fit tests (Table 2.4). In general, the species-specific count models did not cross-validate as well as the logistic regression models (Appendix C).

Patch size and distance to seed source

Burn severity was not a significant factor influencing seedling presence or abundance for any of the species except Douglas-fir. Rather, distance to a live seed source overrode burn severity in the models. Specifically, sites within patches burned at high and moderate severity were further from live seed source trees than sites in either low or unburned patches (Figure 2.4; Kruskal-Wallis Test: $\chi^2 = 117.809$, d.f. = 3, $p << 0.001$). Sites that burned at moderate severity were a median distance of 31 m from a live seed source, while sites that burned at high severity were a median distance of 122 m from a live seed source, compared to 12 and 13 m for unburned and low severity sites, respectively.

Patches burned at higher severity were characterized by high tree mortality, though they tended to be adjacent to patches with only partial tree mortality. The mean distance from an area within a patch burned at high severity to an edge of an unburned or low severity patch ranged from 33 to 118 m for fires that burned in 2000 and from 31 to 122 m in fires that burned in 2007. Over 85% of the area burned in 2000 in high severity patches was less than 95 m from the nearest edge of a lower severity patch and 98% of the area was within 200 m from an edge (Appendix C; Figure C2). Likewise, in the 2007 fire events, 75% of the area burned by high severity fire was within 95 m of an edge and 94% of the area was within 200 m of an edge burned with lower severity (Appendix C; Figure C2). The distribution of edge distances characterized from the satellite data is consistent with the distribution of our on-the-ground measurements of distance to seed source (Appendix C; Figure C2). Although the pixel based and on-the-ground measures of distance to a seed source differ, the distribution of high severity patch sizes we sampled is comparable to the high severity patch size distributions of the satellite data, with 85% of the sites we sampled within 95 m of a live seed source (Appendix C; Figure C2).

Discussion

Resilience of forests to large, severe wildfires is of ecological and management significance, particularly given ongoing climate change and the potential implications of increased forest density from prior forest and fire management (Keeling et al. 2006; Naficy et al. 2010). Our results highlight important interactions between the spatial distribution of patches burned at high-severity and seed dispersal mechanism as the primary controls of post-fire regeneration. Abundant seedling regeneration and the presence of few large high-severity patches across our study region suggest that successful reestablishment is widespread and these dry-mixed conifer forests will be resilient to recent large fires. Furthermore, our results have important implications for addressing the impacts of shifting fire regimes and climate change on forest persistence now and into the future.

Dispersal distance is a primary control on post-fire regeneration

In dry-mixed conifer forests of the U.S. northern Rockies, post-fire regeneration is strongly controlled by landscape structural characteristics resulting from burn patterns. Distance to a live seed source was the primary limitation on post-fire regeneration for all the species across our study region except lodgepole pine, where the presence of serotinous cones allowed for regeneration in the absence of live trees. Burned areas without nearby residual live seed trees had few or no conifer seedlings five to 13 years after fire, regardless of the severity with which the patch burned. Observed post-fire seedling regeneration of Douglas-fir, ponderosa pine, and grand fir mimicked the expected dispersal curves of wind-dispersed species, where the number of viable seeds deposited decreases exponentially with distance from the patch edge (Greene and Johnson 1996). This observed pattern suggests that dispersal limitation is more significant than density-dependent mortality in determining post-fire recruitment across our broad study region (Greene and Johnson 2000).

Our study adds to a growing body of literature emphasizing the importance of nearby live seed sources for post-fire regeneration (e.g., Keyser et al. 2008; Donato et al. 2009; Haire and McGarigal 2010). We identified a 95 m threshold from residual live seed sources for tree seedling establishment to occur, integrating differences in climate, burn severity, and biotic environmental conditions among all 182 sites across our large study region. This threshold falls within the range of dispersal distances documented for many species found on our sites, which can range anywhere from 20 m up to 180 m for ponderosa pine and Douglas-fir and seeds and between 40 and 120 m for the majority of grand fir seeds (McCaughey et al. 1986, Vander Wall 2003).

Because dispersal distance acts as a primary filter on post-fire conifer regeneration, the size and spatial configuration of stand-replacing patches across the landscape become key drivers of post-fire successional trajectories (e.g., Haire and McGarigal 2010). Even within the large wildfires we sampled, more than 75% of the area within patches burned by stand-replacing fire was less than 95 m from an edge, implying that the majority of burned area was close to live seed sources and thus had a high probability of successful natural regeneration. This finding is corroborated by several prior studies from diverse forest types; for example, 75% of stand-replacement patches in subalpine forests in Yellowstone National Park were less than 200 m from a live forest edge (Turner et al. 1994), and 58% of stand-replacement patches in a mixed-conifer forest in southern Oregon were within 200 m of a live forest edge (Donato et al. 2009). Thus, even within large fires, only a small proportion of the entire burned area, including patches burned with high severity, are far enough from seed sources to limit successful natural regeneration. This diversity in burned patch sizes creates landscapes that are largely resilient to mixed-severity fires, regardless of burning conditions or forest type.

Secondary controls on seedling regeneration

Where seed sources were available, seedling regeneration was highly variable, suggesting that seed source is a necessary but insufficient explanation of seedling abundance. Abundant regeneration depends upon successful germination, survival, and growth, all of which are influenced by stochastic processes and environmental conditions that vary at fine scales (e.g., Bonnet et al. 2005). Tree basal area was the primary biotic environmental variable influencing seedling abundance in our study. Basal area can be a proxy for site productivity as it measures both size and density of trees in a stand. Gradients in resource availability that make pre-fire stands productive, such as soil fertility and moisture availability, likely also influence post-fire germination and tree seedling survival (e.g., Clarke et al. 2005; Röder et al. 2008; Casady et al. 2010). Relative to distance to seed source, however, our results suggest only a minor influence of tree basal area on tree seedling regeneration, as all sites with a basal area < 5 m^2 ha^{-1} had a relatively high probability of tree seedling establishment. Therefore, this effect simply indicates that where trees occurred prior to fire, they are likely to regenerate post-fire. Marginal sites with few trees prior to the fire (those with a basal area < 5 m^2 ha^{-1}) are less favorable for reestablishment. Stochastic processes that we did not quantify, such as variability in seed crops, microsite conditions, or favorable post-fire climate conditions (Brown and Wu 2005; League and Veblen 2006), could also account for some of the unexplained variability in seedling abundance across our study region. Seed masting events occur on average every 3 to 12 years in Douglas-fir and ponderosa pine stands in the U.S. northern Rockies (USFS 2012), for example, strongly limiting seed availability in intervening years. Likewise, seed predation likely limits successful germination and subsequent tree seedling regeneration (Vander Wall 1994; Zwolak et al. 2010; Lobo 2014). Given that the post-fire seedling recruitment period can be an important stage within long-term forest succession, further research examining the influence of stochastic variables, especially weather and climate, on post-fire regeneration, will be important for understanding the potential implications of shifts in climate on longer-term forest dynamics.

The importance of environmental gradients was most pronounced for Douglas-fir regeneration. Moderate- and low-severity burns had significantly more Douglas-fir seedlings than unburned sites, suggesting that moderate increases in resource availability, such as increased light, nutrients, or mineral soil, favored seedling establishment and survival (e.g., York et al. 2003, Moghaddas et al. 2008). Recruitment was also more abundant on sites characterized by higher average elevation and lower heat load, representing cooler, wetter locations. High summer temperatures and water availability limit Douglas-fir growth across its range in the northern Rockies (Littell et al. 2008), and our data suggest that these factors also limit seedling establishment and survival. Douglas-fir abundance was also higher on sites with a longer time since fire. For a sporadic seed producing species like Douglas-fir, this is expected, as the probability of successful establishment accumulates over time. However, we caution the extrapolation of these results to sites outside of our study region given that this model did not cross-validate well, nor was this relationship significant for the other species examined. Further, this result suggests that regeneration for the other species we examined may have been pulsed in a single event post-fire, a result of seedling mortality over time, or due to our sample period not being long enough to pick up multiple pulses of successful post-fire establishment.

Post-fire tree seedling composition reflects pre-fire stand composition

Our species composition data show relatively little difference between the species composition of regenerating seedlings and the pre-fire mature tree composition. This suggests that the mix of burn severities, patch sizes, and environmental conditions across the landscape perpetuated the forest conditions that were present prior to fire. At the site scale, post-fire seedling composition may still vary as a function of pre-fire species composition of the live tree edge and species-specific tree regeneration mechanisms. For example, the abundance of seeds reaching the interior of patches depends upon a tree species' dominance on the intact forest edge (Greene and Johnson 2004), where edge dominance increases the seed rain of that particular species (e.g., Greene and Johnson 1996). Infilling by shade tolerant species in the absence of fire could therefore alter the seed rain available to recolonize a patch post-disturbance (Perry et al. 2011), especially if those species are prolific seed producers. These factors, in combination with our model predictions of further dispersal distances for grand fir (i.e., 2-3 times the distance of Douglas-fir and ponderosa pine) and additional empirical evidence (McCaughey et al. 1986), suggest that grand fir has the potential to recolonize larger burned patches and increase in dominance post-fire (e.g., Crotteau et al. 2013). This effect may be especially pronounced in areas that have reduced tree cover of Douglas-fir and ponderosa pine from prior logging.

Species-specific regeneration mechanisms may also determine which dry-mixed conifer tree species recolonize different patches post-fire. Lodgepole pine, for example, can regenerate in the absence of a live seed source where it has serotinous cones stored in an aerial seed bank. Though serotiny can vary considerably across a landscape (Schoennagel et al. 2003), forests with high pre-fire serotiny can have prolific regeneration in stand-replacing patches (Turner et al. 1997; Schoennagel et al. 2003). Compared to the other dry-mixed conifer species we studied, lodgepole pine recruitment is likely to be favored in large high severity (i.e., stand-replacing) patches where live seed sources are limited.

Implications for forest resilience

Our results suggest that the spatial characteristics of mixed-severity fires and pre-fire species composition interact to promote resilience of dry mixed-conifer forests to large wildfires, even when those fires burn under a variety of weather conditions. Most of the area within patches burned by high severity (i.e., stand-replacing) fire in the large regional fire events we studied were close to live trees which likely included seed sources for conifer regeneration. Over 80% of the patches burned at low severity and approximately 40% of patches burned at moderate severity that we sampled exceeded the desired tree seedling densities considered sufficient to regenerate a stand to its pre-fire density, which can range from 180 trees ha^{-1} to 370 trees ha^{-1} in dry mixed-conifer forests across the northern Rockies region (S. Fox, USFS; pers. comm.). Sparse natural tree regeneration was primarily observed in large, high severity patches (i.e., those patches with interiors > 95 m from an edge and few surviving trees). Although burn severity was a poor predictor of ecological response in our study, it is directly related to dispersal distance, as by definition high severity patches have fewer residual live trees and are further from live seed sources. Therefore, the high heterogeneity of patch types and sizes within a fire (i.e., the burn mosaic) is key to maintaining current forest diversity and structure after future wildfires.

Although our seedling regeneration data represent a limited snapshot in time, mortality of seedlings has been shown to decline markedly and remain constant after the year of germination (Pausas et al. 2003, Calvo et al. 2013). Seedlings that establish and survive the first year of growth are likely to remain an important feature of long-term forest structure. In our study, 94% (n = 2226) and 72% (n = 2518) of the seedlings that we sampled that regenerated after the 2000 and 2007 fires, respectively, were greater than one year old. Long-term studies of Douglas-fir forests in the western Cascades, USA, indicate that density dependent mortality does occur, especially with canopy closure, after about 25 – 32 years post disturbance. However, decreases in stem density are offset by changes in biomass, where biomass loss is maintained at a relatively low and constant level (Lutz and Halpern 2006). Although continued seedling recruitment and mortality is likely to occur in the stands we sampled, the combination of high seedling densities and large proportions of well established seedlings suggests that if current conditions remain stable, these seedlings are likely to persist.

Although there is considerable concern surrounding large fire events, patch scale heterogeneity present across the burned landscape in the U.S. northern Rockies suggests that these forests will recover to pre-disturbance species composition and diversity. Our conclusion is predicated upon several things: significant seedling mortality does not occur, forests remain unburned long enough for live trees to reach reproductive maturity, the proportion of high-severity patches far from seed sources does not increase significantly in the future, and climate change does not shift post-fire environmental conditions so as to limit successful tree seedling establishment and growth. The range of suitable climate conditions for growth of mature tree species in the U.S. northern Rockies may shift considerably in the coming decades (Rehfeldt et al. 2006; Rehfeldt et al. 2008), and it is likely that the regeneration niche of seedlings is even narrower (Grubb 1977; Jackson et al. 2009). Additionally, extrapolations of statistical fire-climate relationships suggest a potential two to five-fold increase in the median area burned in the U.S. northern Rockies by mid-century (Littell 2011), implying an increase not only in frequency, but also fire size. Short fire-return intervals may limit the potential for tree regeneration success and remove future seed sources (Keeley et al. 1999; Johnstone and Chapin 2006; Brown and Johnstone 2012). More intense and severe fire may also favor tree species that are well adapted to regenerate in the absence of live seed sources nearby (e.g., lodgepole pine and grand fir).

Understanding how large, mixed-severity fires impact the regeneration and resilience of forests will become increasingly important for making sound forest and fire management decisions in a warmer, more fire prone future. Conifer regeneration and habitat restoration are important management priorities following fire, and abundant natural post-fire tree regeneration may limit the area managers need to treat to meet these objectives. In large, high severity patches, sparse natural regeneration may result in delayed successional trajectories or altered vegetation states. Managers aiming to insure post-fire recovery should therefore focus regeneration efforts on areas within high-severity patches that are far (> 100 m) from live seed sources. As area burned continues to increase, the amount of area burned severely will also increase (Dillon et al. 2011), If the size of high severity patches and the relative proportions they occupy on the burned landscape increases with future climate change, or if the post-fire environmental conditions shift significantly relative to the past several decades (Rehfeldt et al. 2006), the resilience of dry mixed-conifer forests to large wildfires that we documented will be increasingly compromised.

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Table 2.1. Predictor variables included in all statistical models and their methods of measurement. For comparative purposes, all variables deemed independent were used in both analyses.

Category	Variable	Method of measurement	Units	Type	Range
Fire	Distance to the nearest live seed source	Field measured (10 closest trees; averaged)	m	Continuous	Bounded [0 500]
	Burn Severity	RdNBR and field verified	Unitless	Categorical	Unburned [0], Low [1], Moderate [2], High [3]
	Time Since Fire	Derived from year of sampling minus year of burn; if unburned during year of sampling, MTBS data from 1984 - present were used to determine prior burn severity and TSF	Yrs	Continuous	Discrete values [5,6,12,13, 19, 26, 29]
	Distances > 500 m	Indicator variable for censored values where a value of [1] indicates the observation was censored (i.e., undetectable) and a value of [0] indicates the variable was not censored (i.e., measured)	Unitless	Indicator Variable; Constant	Discrete [0 or 1]
Abiotic Environment	Elevation	Field measured	m	Continuous	Bounded [675 2203]
	Heat Load Index	Derived from slope, aspect, latitude (McCune and Keon 2002)	Unitless	Continuous	Not bounded [0 1.006]
Biotic Environment	Tree Basal Area	Field measured (3 plots at 0, 30, 60 m; averaged)	m^2/ha	Continuous	Not bounded [0 49.3]
	Understory Vegetation Cover	Field measured (3 plots at 0, 30, 60 m; averaged)	%	Continuous	Not bounded [18.67 148.33]
	Canopy Cover*	Field measured (3 plots at 0, 30, 60 m; averaged)	%	Continuous	Bounded [0 100]

* Canopy cover was not included in any analysis because it was significantly correlated ($\rho > 0.5$) with distance to seed source and stand density.

Table 2.2. Density, richness, and species composition of pre-fire trees and post-fire seedlings. Conifer species composition is calculated as the percent of total tree density pre-fire and seedling density post-fire for on each site, averaged across all sites. Listed under each species in parentheses is the number of sites on which each species was found (n = 182). Western white pine was found on one site but is not included in the table because of its low abundance. Data are means +/- 2 SE.

Time	Conifer Density (trees ha^{-1})	Average Conifer Species Richness (by plot)	Conifer Species Composition* (Mean % of plot density ± 2 SE)						
			Douglas-fir (120)	ponderosa pine (58)	lodgepole pine (48)	grand fir (27)	Englemann spruce (22)	western larch (13)	subalpine fir (11)
pre - 2000 fire (n = 84)	361 ± 61	1.8 ± 0.1	38.9 ± 3.4	12.3 ± 4.2	29.1 ± 12.9	20.3 ± 9.4	NP	19.8 ± 19.5	NP
post - 2000 fire (n = 84)	7047 ± 1714	2.1 ± 0.1	35.3 ± 4.3	16.2 ± 7.1	19.2 ± 7.9	24.1 ± 10.6	2.5 ± 2.3	16.9 ± 13.1	13.1 ± 12.0
pre- 2007 fire (n = 98)	421 ± 48	2.2 ± 0.1	32.7 ± 3.9	18.9 ± 5.0	15.8 ± 5.6	22.3 ± 9.7	6.8 ± 7.3	7.1 ± 6.6	18.5 ± 6.2
post- 2007 fire (n = 98)	8153 ± 2006	2.2 ± 0.1	24.1 ± 4.4	23.1 ± 5.6	24.0 ± 5.7	19.0 ± 7.5	5.1 ± 4.6	9.3 ± 11.8	1.7 ± 1.7

* Western white pine was found on one site but its density was so low that it did not contribute to the overall species composition. NP indicates that species was not present on any of the sampled sites.

Table 2.3. Parameter estimates from the logistic and negative binomial models for each predictor variable. Estimates are only listed if they were significant for that model. Asterisks indicate levels of significance: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. For each species-specific model, a censored variable was included for distance to a live seed source, where 1 indicates that the distance was accurately measured, and a 0 indicates that the distance was > 500 m or not detected visually from the transect center. If significant, the censor variable indicates that the odds of seedling abundance were lower without a live seed tree present.

	All seedlings		Douglas-fir		ponderosa pine		lodgepole pine		grand fir	
	binary	count	binary	count	binary	count	binary	count	binary	count
Distance to Seed Source (m)	-0.014**	-0.007***	-0.011**	-0.006**	-0.011*	-0.009*	-	-	-0.013*	-
Low Burn Severity	-	-	-	2.051*	-	-	-	-	-	-
Moderate Burn Severity	-	-	-	2.578**	-	-	-	-	-	-
High Burn Severity	-	-	-	-	-	-	-	-	-	-
Time Since Fire (yrs)	-	-	-	0.146***	-	-	-	-	-	-
Censor Variable (Dist. to Seed Source)	NA	NA	-	-1.204*	-2.630***	-3.085***	-3.134***	-2.252**	-2.701***	-2.049**
Elevation (m)	-	-	0.001*	0.001*	-	-	0.002*	-	-	-
Heat Load Index	-	-	-2.452*	-2.997***	-	-	-2.271*	-	-	-
Tree Basal Area (m^2/ha)	0.072**	0.032*	0.072**	0.042**	-	-	-	-	-	-0.070*
Understory Vegetation Cover (%)	-	-	-	-	-	-	-	-	-	-0.043***
Log(Area)	NA	-1.147***	NA	-1.020***	NA	-	NA	-1.505*	NA	-0.895*

Table 2.4. Performance among the full logistic and negative binomial models for all species and the four most abundant species. Model fit statistics indicate a significant lack of fit of the distribution to the data if the p-values are < 0.05. PPR is the positive predictive rate, defined as the proportion of times presences were correctly predicted as such. NPR is the negative predictive rate, defined as the proportion of times absences were correctly predicted as such.

	Logistic Regression								Negative Binomial GLM		
	Deviance residual		Hosmer-Lemeshow statistic		Reciever Operating Curve (ROC)				Spearman's rank	Pearson's statistic	
Models	χ^2	p	χ^2	p	AUC [95% CI]	Classifier	PPR	NPR	ρ	χ^2	p
All species	156.4	0.781	4.7	0.789	0.826 [0.758 - 0.894]	0.87	93.2	39.4	0.70	1066.5	0.144
Douglas-fir	165.6	0.601	13.7	0.091	0.850 [0.790 - 0.909]	0.84	91.4	46.8	0.71	455.2	0.337
ponderosa pine	166.1	0.591	18.5	0.018	0.841 [0.782 - 0.900]	0.66	83.3	75.9	0.49	122.3	0.007
lodgepole pine	150.3	0.871	7.3	0.503	0.857 [0.795 - 0.919]	0.65	80.0	80.2	0.60	1287.8	0.132
grand fir	108.3	0.999	10.0	0.263	0.847 [0.765 - 0.929]	0.58	68.8	91.0	0.36	329.8	0.127

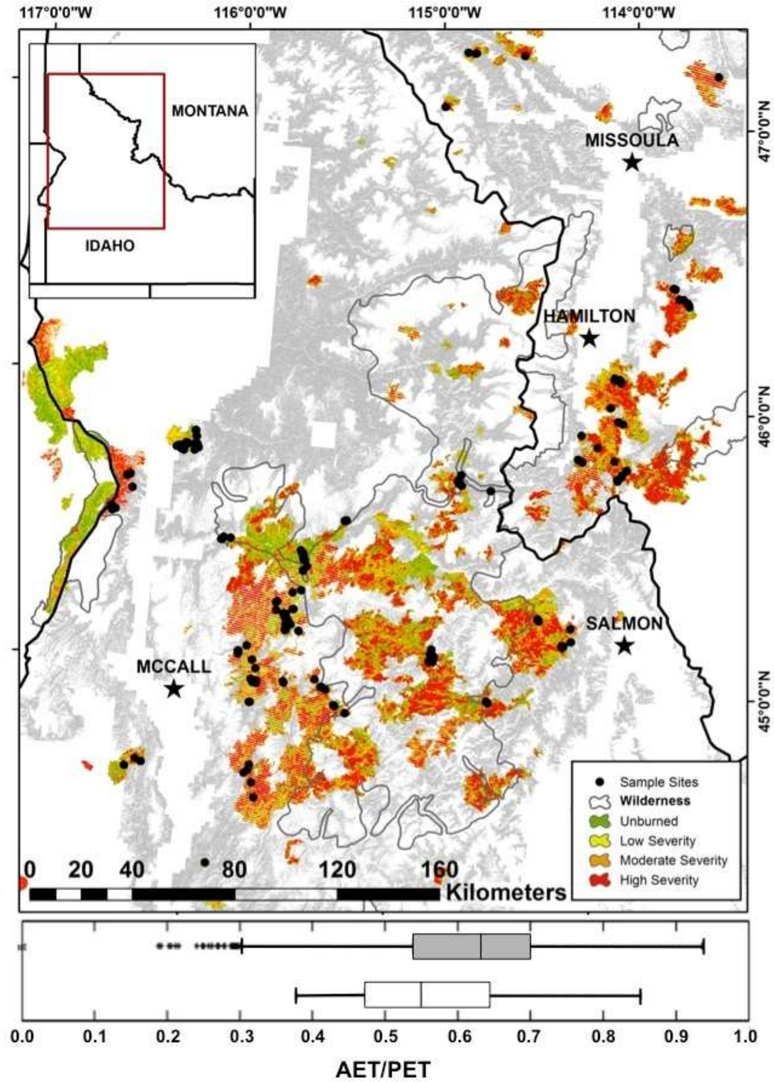

Figure 2.1. We sampled 182 sites in 21 individual fire events that burned in either 2000 or 2007 across central Idaho and western Montana. We stratified sites across gradients in elevation, aspect, and burn severity to represent the full range of climates in dry-mixed conifer forests (grey shading). Climate is defined here by the ratio of actual to potential evapotranspiration (Appendix B), shown below the map with boxplots for dry-mixed conifer forests (grey) and our sample sites (white). Boxplots delineate the 25th, 50th, and 75th percentiles and whiskers correspond to the 10th and 90th percentiles.

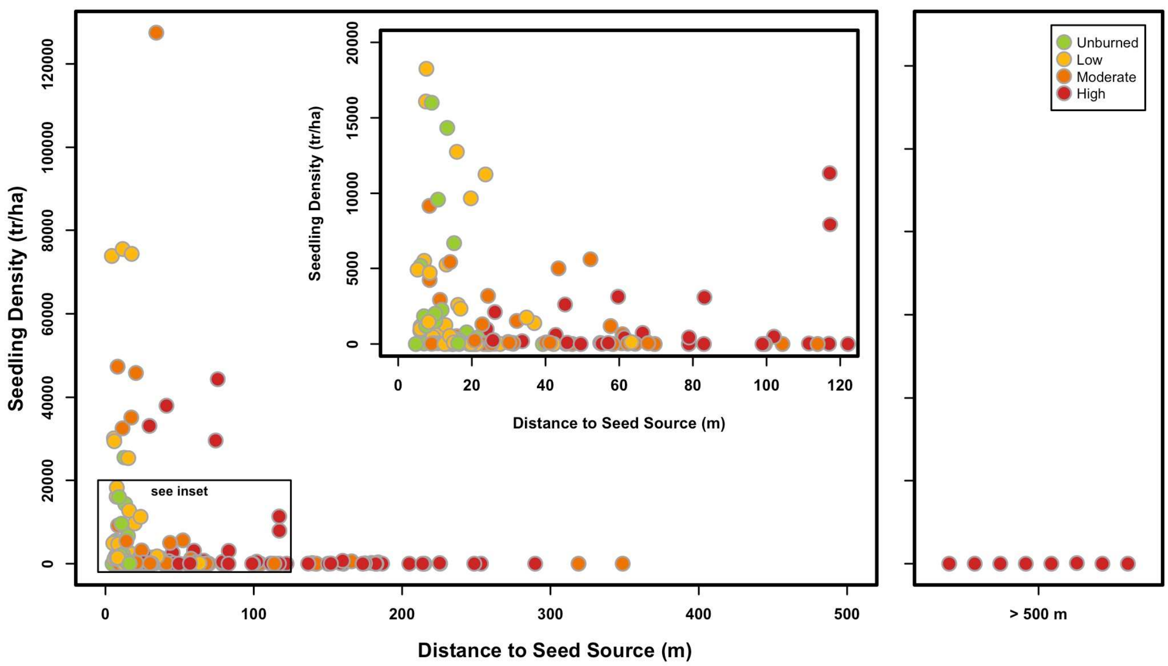

Figure 2.2. Tree seedling density as a function of the distance to a live seed source. Each site is additionally colored by burn severity. The maximum distance that could be measured was 500 m from the transect center. Sites with no live seed sources within 500 m are presented in the right panel.

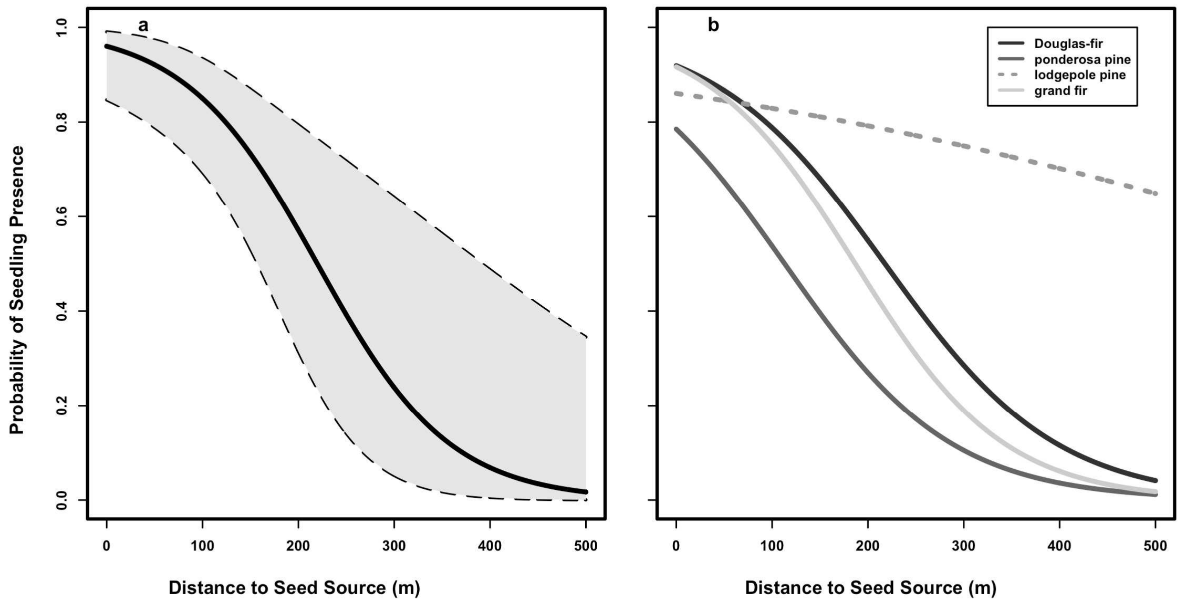

Figure 2.3. Logistic regression model results. Relationship between the probability of seedling presence and distance to a live seed source for (a) all species and (b) the four most abundant species, when all other variables in the model are held at their median values. The shaded region between the dotted lines represents the 95% confidence intervals on the predicted values for the all species model. In panel (b), the dashed line indicates that the relationship between distance and seedling presence was not significant for that species ($p > 0.05$). Confidence intervals are not shown in panel (b) because they overlap for all species.

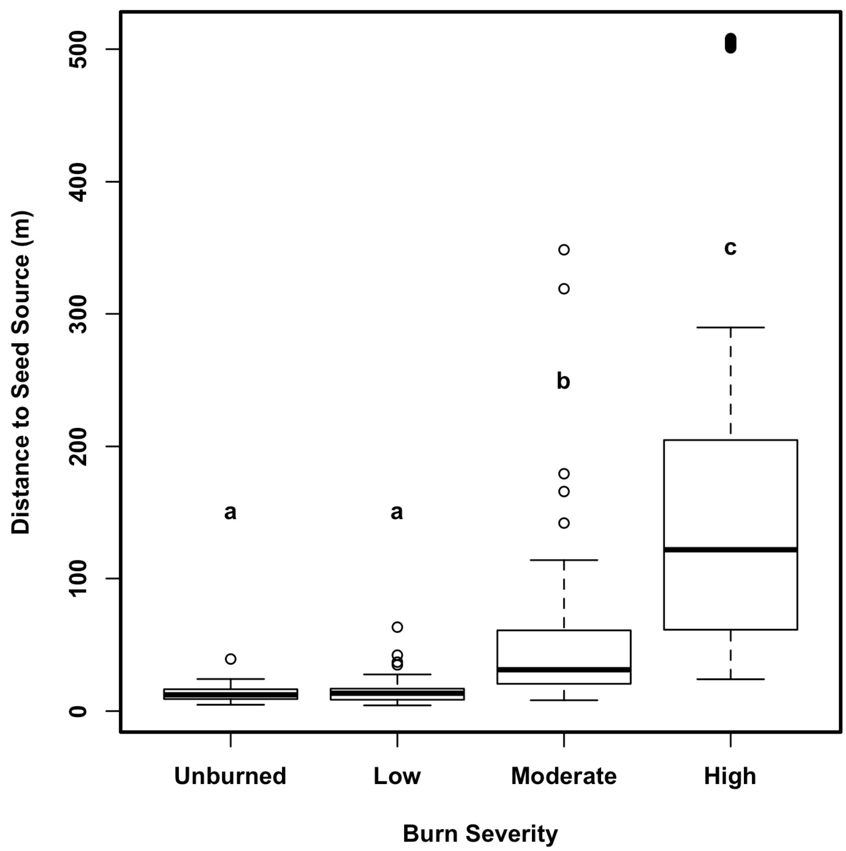

Figure 2.4. Relationship between burn severity and the average distance to a live seed source. Burn severity was initially categorized using satellite RdNBR data and subsequently field verified. Letters denote to statistical differences between the treatments. Patches burned at high severity have significantly further average distances to live seed trees than moderate, low, or unburned patches. Patches burned at low severity did not differ from unburned patches in the average distance to a live seed source.

CHAPTER 3

Interactions Between Climate and Fire Indicate Sensitivity of Post-Fire Tree Regeneration to Future Climate Change

Kerry B. Kemp • Philip E. Higuera • Penelope Morgan • John T. Abatzoglou

Abstract

Climate change is predicted to shift the distribution and occurrence of plant species in the future, and in many ecosystems shifts will also be mediated in part by broad-scale disturbances that influence mortality and recruitment patterns across landscapes. Ecosystem dynamics after disturbances, such as tree recruitment after wildfires, will therefore be an important determinant of future change. To understand the interactions among wildfire, climate, and ecosystem recovery, we studied post-fire tree seedling regeneration across 177 dry mixed-conifer forest sites burned in 2000 and 2007 in the Rocky Mountain region of central Idaho and northwestern Montana (Northern Rockies). We used generalized additive models to quantify how the density of Douglas-fir and ponderosa pine seedlings varied as a function of climatic (temperature, precipitation, soil moisture, and evapotranspiration) and fire-related variables (distance to live seed sources, live tree stand density, and burn severity inferred from satellite imagery). Both climate and fire modified post-fire regeneration of Douglas-fir and ponderosa pine, with summer temperature particularly influential in our statistical models. Specifically, seedling abundances were greatest where mean summer temperatures were between 11.2 - 17.5 °C for Douglas-fir and 12.3 - 18.5 °C for ponderosa pine. Douglas-fir regeneration was also higher when live seed sources were less than 100 m away. While seed source limitations impact whether seeds can successfully disperse into a burned area, climate overrides the influence of live seed sources once mean summer temperatures exceed a threshold of approximately 17 - 18°C. Using downscaled climate projections from the Intergovernmental Panel on Climate Change's AR5, representative concentration pathway 8.5, 82% of the sites we sampled in dry mixed-conifer forests are projected to exceed the 17°C temperature threshold for tree seedling establishment by mid-century, with seedling densities of Douglas-fir and ponderosa pine projected to decrease on 77% and 59% of sites where they are currently found. In response to warming climate, ponderosa pine and Douglas-fir will likely shift upwards in elevation in the future. The combined impacts of climate and fire will be evident where tree mortality is extensive due to future fires and post-fire regeneration is lacking due to unsuitable climate conditions for reestablishment.

Keywords Climate change • burn severity • mixed-severity wildfire • tree regeneration • range shifts

Introduction

Climate change and warming temperatures have been well documented globally over the past century (IPCC 2013). Climate serves as an important driver of vegetation patterns and variation in forest species composition has been well established (e.g., Whittaker 1960, Woodward 1987, Stephenson 1990, Davis and Shaw 2001). As 21st century climate projections suggest continued increases in temperature and regional and seasonal changes in precipitation (IPCC 2013), the distribution of many forest species is consequently projected to shift (e.g., Rehfeldt et al. 2008, Gonzalez et al. 2010). Although climate is implicated as a primary driver of forest change, these changes will likely be mediated in part by complex interactions between climate and disturbance.

In many temperate coniferous forests, disturbances like wildfire play a critical role in forest development by affecting tree mortality and recruitment. Wildfires create opportunities for recruitment by increasing suitable sites or resource availability for tree regeneration (Ehle and Baker 2003, Schoennagel et al. 2011, Tepley et al. 2013). For example, fire creates patches of bare mineral soil, increases light and nutrient availability and reduces competition from surrounding vegetation (York et al. 2003, Gray et al. 2005, Moghaddas et al. 2008). These opportunities are modified by life history traits of individual tree species and heterogeneity in fire patterns (Donato et al. 2009, Haire and McGarigal 2010, Kemp et al. 2015a). For example, the severity and spatial configurations of burned patches will impact dispersal and availability of seeds (Savage and Mast 2005, Haire and McGarigal 2010), constraining regeneration in some areas while creating ample opportunities for establishment in other areas. Where fires kill established trees, a lack of post-fire recruitment could result in punctuated shifts in species composition or a landscape-scale conversion from forest to non-forest vegetation, making the post-fire recruitment phase of forest development an important indicator of future change.

While many western U.S. tree species evolved with fire, changing fuel conditions in low elevation dry forests, in part resulting from past land management practices such as logging and fire exclusion (Keeling et al. 2006, Naficy et al. 2010), in combination with more extreme climate conditions (e.g., longer fire seasons, low fuel moisture; Jolly et al. 2015), have been implicated in recent changes in patterns of wildfire (Higuera et al. 2015). For example, over the past century, increased fire season length and large fire occurrence across the western U.S. has been linked with increasing global and regional mean temperatures (Jolly et al. 2015), warmer springs and earlier snowmelt (Westerling et al. 2006), and drought (Abatzoglou and Kolden 2011, Dennison et al. 2014). Similarly, increases in burn severity across dry forests have been observed in regions of the west, where exceptionally dry conditions during the fire season overwhelm other controls (e.g., topography) of burn severity (Miller et al. 2009, Dillon et al. 2011, Miller et al. 2011, O'Connor et al. 2014). Strong linkages between climate and fire (e.g., Westerling et al. 2003, Morgan et al. 2008, Littell et al. 2009, Parisien et al. 2012, Higuera et al. 2015) implicate an increasing probability of more extensive, and potentially more severe, fires in a warmer future (Flannigan et al. 2009, Liu et al. 2010, Littell 2011, Barbero et al. 2014).

Forest response to future climate change will therefore reflect the impacts of both changing disturbance regimes and changing climate, and these changes are likely to be most evident along the margins of an existing ecotype, such as low elevation dry mixed-conifer forests. Assessing how changes in the frequency, extent, and severity of fires will affect future forest ecosystems is difficult, however, because of complex interactions among vegetation and climate at local and regional scales. Bioclimatic vegetation models use climate variables characterizing modern distributions to anticipate where vegetation will exist in the future based on climate projections (e.g., Pearson and Dawson 2003, Rehfeldt et al. 2006, Botkin et al. 2007) without taking into account other biotic and abiotic constraints, such as dispersal potential, disturbance and competition. As fires become more extensive, post-fire mortality and regeneration will play a critical role in determining the extent and location of changes in tree species ranges. For example, the establishment of tree seedlings after fire will be constrained by habitat suitability and environmental conditions (Lenoir et al. 2009).

Young trees have a narrower climatic tolerance than their adult counterparts (Jackson et al. 2009, Dobrowski et al. 2015), and therefore forest composition preceding a fire may be a poor predictor of future composition under a changing climate. Additionally, changes in species distributions are likely to be modified by local topography and microclimate (Dobrowski 2011). As future growing season climate becomes warmer and drier (Mote & Salathe 2010), regeneration in western U.S. forests may shift towards more mesic microclimates, including north-facing slopes (Donnegan and Rebertus 1999) and to higher elevations (Dodson and Root 2013).

Though it has been hypothesized that warming climate and larger, more frequent fires will impact the distribution and abundance of forest vegetation (e.g., Littell et al. 2010, Serra-Diaz et al. 2015), few studies document the interactions between climate and fire, making it difficult to anticipate the feedbacks among climate, disturbance, and forest change in the future. In order to understand the interacting impacts of climate and disturbance on forest regeneration, we used downscaled climate projections and burn severity metrics to predict the density of ponderosa pine and Douglas-fir seedlings measured after 21 large fires across 177 dry mixed-conifer forest sites in the U.S. northern Rockies. Quantifying the interactions between climate and fire allowed us to identify important thresholds governing seedling regeneration after fire and predict how these interactions might impact future forest recovery.

Methods

Field sampling

Our study region encompasses the dry mixed-conifer forest zone of the U.S. northern Rockies, from north of the Snake River plain in Idaho to south of Glacier National Park in

Montana (Figure 3.1). Steep river valleys bisect the terrain throughout the study region, creating large elevation gradients with distinct vegetation communities on contrasting aspects. South-facing slopes, especially at low elevations are predominately a mix of canyon grasslands dominated by Idaho fescue (*Festuca idahoensis* Elmer) and arrowleaf balsamroot (*Balsamorhiza sagittata* (Pursh) Nutt.) with sparse ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson). At higher elevations and on north-facing slopes, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco)), grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), and lodgepole pine (*Pinus contorta* Douglas ex Loudon) dominate. The study region experiences dry summers and wet winters. Based on downscaled climate data (Abatzoglou 2013), average July temperature across our study sites ranges from 12.0°C at high elevation sites in central Idaho to 22.4°C at low elevation sites in Hells Canyon and average January temperature ranges from -7.0°C to -0.2°C, total annual precipitation ranges from 450 mm to 842 mm. Soils are coarse textured inceptisols and entisols derived from granitic Idaho batholith parent material with little soil water holding capacity (USDA 2014).

We used a stratified random selection procedure to identify sites within dry mixed-conifer forest types, as classified from 30-m resolution LANDFIRE existing vegetation layers (LANDFIRE 2010), and which fell within four burn severity classes (unburned, low, moderate, and high severity; MTBS 2011) in 21 large (> 400 ha) fires that burned in 2000 and 2007, both years of widespread fires in this region (Morgan et al. 2008, Morgan et al. 2014). We inferred burn severity using Relative differenced Normalized Burn Ratios (RdNBR; Miller and Thode 2007) calculated by Dillon et al. (2011) from 30-m Landsat TM+ satellite imagery. We sampled 177 sites during the summers of 2012 and 2013 from the range of burn severity classes, northeast and southwest aspects, and at low, mid-, and high elevations (range

= 675 - 2203 m above sea level). By stratifying site selection across a range of elevations and aspects, our sites represent the climate conditions indicative of dry mixed-conifer forests at and above lower treeline in the U.S. northern Rockies (Figure 3.1).

Detailed field methods are described by Kemp et al. (2015a), and here we summarize our approach. At each site, we counted live tree seedlings on a 60-m long transect that ranged from 1 m to 10 m in width. The width of the transect on each site was determined prior to sampling based on an ocular estimate of tree seedling density; sampling effort was increased up to the maximum transect width in order to sample at least 30 seedlings of the most common species on each site. We also measured canopy cover and stand density of live and dead overstory trees at three locations along each transect and the average distance to ten nearest live seed trees from the transect center.

Climate data

To represent the variability in climatology across our study area, we used a 30-arc second (~ 800-m) resolution gridded climate surface of monthly temperature, precipitation, and dewpoint temperature from the Parameter-elevation Regressions on Independent Slopes Model (PRISM; Daly et al. 2008) averaged for the period 1981-2010. PRISM data are spatially interpolated from weather station observations using known physical relationships to account for differences in factors such as elevation, topographic position, and orographic effects (Daly et al. 2008). Additional variables of wind speed and solar radiation needed for computing reference potential evapotranspiration (PET) using the Penman-Montieth equation (Allen et al. 1998) were acquired from the surface meteorological dataset of Abatzoglou (2013) at 2.5 arc minute (~4-km) grid and bilinearly interpolated to the PRISM grid.

Evapotranspiration for PET was estimated using a standard reference crop and represents the potential amount of evaporative water loss from a site (Allen et al. 1998).

In addition, we used a modified Thornthwaite water balance model (Willmott et al. 1985, updated by Dobrowski et al. 2013) to estimate monthly actual evapotranspiration (AET) and soil moisture, averaged over the 30-year period from1981-2010 at 30-arc second resolution. AET is a measure of the simultaneous amount of water that is available to plants and the energy demand on those plants from the environment (Stephenson 1990), which has proven useful in predicting tree species distributions in other studies (Lutz et al. 2010, Dobrowski et al. 2015) and should, in theory, have better potential to predict variability in seedling regeneration. For this type of water balance model, any water that is not stored in either the soil or snowpack, transpired by vegetation, or evaporated from the ground surface is considered runoff. The calculation requires information on latitude, mean monthly temperature, precipitation and PET, solar insolation, and available soil water holding capacity. We assumed a depth to restrictive soil layer of 250-mm and used available water storage data extracted from the Natural Resource Conservation Service 1-km resolution State Soil Geographic Database (Schwarz and Alexander 1995) to estimate soil water holding capacity. Finally, we estimated water deficit as the difference between PET and AET, which can be interpreted as a measure of drought stress on plants. By incorporating both the availability of water and energy inputs, water balance variables, such as AET, PET, and water deficit may be mechanistically closer to factors that impact plant survival, distribution and abundance.

To model potential future seedling densities, we used two additional climate data sets downscaled from climate projections for 20 climate models from the fifth Coupled Model Intercomparison Project (CMIP5) for both the historical (1971-2000) and mid-century (2041-

2070) periods. We considered projections for climate models runs using the Representative Concentration Pathway (RCP) 8.5 experiment. Data were statistically downscaled using the Multivariate Adaptive Constructed Analogs method (Abatzoglou and Brown 2011) to a 2.5 arc minute (~4 km) resolution and averaged across the 20 climate models (Appendix E, Figure E1). We calculated the differences between mid-21st century projections and historical model runs. To account for spatial differences between the downscaled (4 km) and observed data (800 m), we spatially interpolated and incorporated differences from downscaled climate projections to adhere to the resolution of the observed data. We also applied a correction factor of 0.15°C to the 1971-2000 climate projections to account for the 10 year difference with the observed 1981-2010 data. We used climate data at voxels co-located with each of our 177 sample sites.

Statistical analysis

We used generalized additive models (GAMs) to explore the relationships between climate, fire, and post-fire seedling abundance of the two most common species encountered on our sites: Douglas-fir and ponderosa pine. GAMs are an ideal modeling tool for our questions because we had no expectations of a linear relationship between predictor and response variables (e.g., seedling establishment might increase at moderate levels of burn severity but be less at lower and higher burn severities). We considered a suite of derived temperature, precipitation, and water balance variables related to growing season conditions with mechanistic links to seedling germination and establishment including antecedent precipitation and annual water deficit (Table 3.1). We also chose variables that represented post-fire conditions that were likely to affect the post-fire microenvironment and burned patch size (Table 3.1). We included the logarithm of sampling area in each of the models to account for the variable transect sampling scheme.

We developed a suite of candidate models that included one variable from each category (Table 3.1, Appendix D, Table D1). We tested each of the explanatory variables for collinearity and excluded any variables that were well correlated (i.e., $|r| > 0.6$) from our candidate models. Fourteen possible models were constructed for both Douglas-fir and ponderosa pine, and each of the candidate models was fit with a negative binomial distribution (Appendix D, Table D1), which accounted for the high overdispersion present in the data. Of the 14 candidate models, we selected a final model based on AIC, explained deviance and residual plots (Appendix D, Table D1), and we tested for assumptions of concurvity following the procedures outlined in Zuur et al. (2009).

To minimize dispersion in model residuals, we removed outliers and log transformed variables displaying non-normal residual patterns. We also checked for spatial and temporal correlation among the sites in our data and added random effects to account for the different residual variance among plots (Zuur et al. 2009), depending on the year of sampling (referred to herein as time since fire) or the fire event that the plots were sampled within. All potential interaction terms were added to each model one at a time to see if they improved model fit. Interaction terms that improved model fit were added to the final model.

We developed each of our final candidate models with all possible explanatory variables and interaction terms and subsequently reduced the model with a backwards-stepwise selection procedure using AIC and a Chi-squared analysis of deviance to assess model fit until the model contained only significant terms (Zuur et al. 2009). The addition of random effects and interaction terms led to an improved overall AIC and explained deviance with the least amount of possible terms for both species (Table 3.2). Final models were implemented with a tensor product smoothing function (see Hastie and Tibshirani 1990, Wood 2006).

Models were developed using current climate data and observed tree seedling counts measured at each of our sampled sites, and then we applied the models to predict current and future seedling counts using future climate projections. To test model reliability for making accurate predictions outside of the calibration dataset, we compared the sampled seedling densities to those predicted by the models using Spearman's rank correlation (Table 3.2). We cross-validated the models by randomly withholding 75% of the data to train the model and used the remaining 25% of the data for model validation for each of 1000 bootstrapped model runs. Cross-validation statistics and the Spearman's rank correlation coefficient were summarized from each of the bootstrapped samples (Table 3.2). We quantified projected future increases and decreases in seedling densities at each of our sites by comparing predicted seedling densities from the initial parameterized models to densities predicted using future climate data, expressed as a percentage of change (future - historical / historical). Finally, to identify where changes in species abundance are most likely to occur given our future scenarios, we compared the elevation distribution of sites with predicted increases and decreases in seedling density to the elevation distribution of sampling sites where seedlings of these species were present (i.e., 1 or more seedlings on site).

Results

Of the 177 sites sampled in 21 fires, 64% had Douglas-fir and 31% had ponderosa pine seedlings present. Although seedling densities varied widely (e.g., Douglas-fir: 0 - 68000 trees ha^{-1}; ponderosa pine: 0 - 1833 trees ha^{-1}), we were able to explain over 50% of the variability in the density of Douglas-fir and ponderosa pine seedlings across our study area using climate and fire variables. Cross-validation and a chi-squared goodness of fit tests confirmed the fit of a negative binomial distribution to our data (Table 3.2), and relatively high correlation between the predicted and observed seedling densities for both species suggested the models' ability to accurately predict Douglas-fir ($\rho = 0.81$) and ponderosa pine ($\rho = 0.50$) seedling densities (Table 3.2).

Summer temperature and distance to a live seed source were most important for determining seedling density. Our initial best-fit statistical model for Douglas-fir tree seedling density included summer temperature, AET, distance to a live seed source, burn severity and live tree stand density (AIC = 1127.56; Appendix E, Table E1). All variables were included in the model as concurvity was below 0.5. Residual patterns and AIC values were improved with the addition of random effects for both time since fire, fire event and an interaction term between summer temperature and distance to a live seed source. Our final reduced model included only summer temperature, log of the distance to a live seed source, live tree stand density, random effects, and the interaction between summer temperature and distance to a live Douglas-fir seed source (AIC = 1088.10; Appendix E, Table E1). Both the log of distance to seed source and live tree stand density were included as parametric terms in the final model as both had an estimated degrees of freedom of one for the smoothing parameter, suggesting a linear relationship between each of the explanatory variables and the response.

Estimated counts of Douglas-fir seedlings decreased with live tree stand density ($\beta = 0.00$, $p = 0.01$; Figure 3.2a) and with distance to live seed source ($\beta = -1.06$, $p < 0.001$; Figure 3.2a). Mean summer temperature also significantly influenced Douglas-fir seedling counts.

The highest probabilities of Douglas-fir seedling densities, defined by the central 95% of the partial dependence plot, occurred between 11.2 and 17.5°C, with a maximum probability around 15°C (Figure 3.2c).

Summer temperature remained an important variable for predicting ponderosa pine densities. The initial best fit model for ponderosa pine seedling density included the ratio of actual to potential evapotranspiration, summer temperature, distance to a live ponderosa pine seed tree, burn severity and canopy cover (AIC = 554.81; Appendix E, Table E2). No variables displayed significant concurvity and thus all were kept for the model selection process. After stepwise reduction, only one main effect, summer temperature, remained significant in the ponderosa pine model and no interaction terms were deemed to significantly improve model fit (AIC = 529.89; Appendix E, Table E2). The model also included both time since fire and fire event as random effects. Although time since fire was not significant in the model, it could not be eliminated from the model without worsening model fit. The relationship between summer temperature and ponderosa pine seedling count was similar to that of Douglas-fir with ponderosa pine seedling counts increasing up to a mean summer temperature of 14.8°C (Figure 3.2d). However, ponderosa pine seedlings were tolerant of warmer summer temperatures than Douglas-fir seedlings and 95% of the predicted sites where seedling densities were greater than zero had summer temperatures between 12.3 and 18.5°C (Figure 3.2d).

The interaction between summer temperature and distance to a live seed source underscores the importance of dispersal distance in predicting post-fire patterns of Douglas-fir regeneration in dry mixed-conifer forests under current climate conditions. For example, based on our statistical model for Douglas-fir, predicted seedling densities vary strongly as a function of temperature when distance to seed source is less than approximately 50 - 100 m; in contrast, when distance to seed source is greater than approximately 100 m, our model predicts lower seedling densities regardless of temperature (Figure 3.3a). Likewise, our model reveals a temperature threshold, beyond which the distance to seed source has little influence on predicted Douglas-fir abundance; above approximately 17 - 18°C, predicted Douglas-fir seedling densities are low (< 90 trees ha^{-1}) regardless of the distance to seed source (Figure 3.3a). This effect is less pronounced for ponderosa pine (Figure 3.3b), though seedling densities are also predicted to be low on sites with a mean summer temperature above approximately 19°C. Currently, only 19% of sites that we sampled in the dry mixed conifer forest cover type have a mean summer temperature exceeding 17°C. However, 82% of the sites we sampled are projected to have a mean summer temperature greater than or equal to 17°C by the middle of the 21st century, based on the RCP 8.5 projections (Figure 3.3c).

Informing our models with future climate data highlights the potential degree of change in seedling regeneration after future fires in different locations across the landscape. Although 82% of sites are projected to have mean summer temperatures greater than 17°C, our models for seedling density project that 77% of our sample sites will experience at least a 1% decrease in Douglas-fir tree seedling regeneration in the future (compared to current models of seedling density), while only 22% of sites are projected to support increased seedling densities (Figure 3.4a). Ninety percent of sites with mature Douglas-fir had tree densities ranging between 2 and 754 trees ha^{-1} (median = 50 trees ha^{-1}) prior to the fire, yet 90% of future Douglas-fir seedling densities are predicted to be less than 294 trees ha^{-1} (median = 15 trees ha^{-1}). In contrast, 31% of sampled sites showed no change in predicted future ponderosa pine seedling densities while on approximately 10% of sites ponderosa pine seedling densities were projected to increase, and the remaining 59% of sites densities were projected to experience decreased seedling densities (Figure 3.4b). Mature ponderosa pine densities ranged up to 181 trees ha^{-1} (median = 0 trees ha^{-1}) prior to the fire and future seedling densities are projected to be up to 126 trees ha^{-1} (median = 15 trees ha^{-1}).

Given the strong dependence of seedling densities on temperature, sites projected to experience an increase in seedling densities will occur at higher elevations than sampled sites where ponderosa pine and Douglas-fir are currently found (Figure 3.5). The median elevation of sites where Douglas-fir are currently present is 1572 m, while the majority of sites where Douglas-fir densities are predicted to increase by mid-century will have a median elevation of 1727 m, a difference of 155 m (Figure 3.5a). Likewise, the median elevation of sites where ponderosa pine seedlings were present during sampling was 1487 m (Figure 3.5b). The median elevation of sites where ponderosa pine densities are projected to increase is 1833 m, a difference of 346 m (Figure 3.5b). Seedling densities are projected to decrease in the future at similar elevations to where Douglas-fir and ponderosa pine are currently present (Figure 3.5).

Discussion

Our results underscore the importance of both fire and climate in influencing post-fire seedling regeneration in dry mixed-conifer forests, but they further allow us to assess the relative importance of each factor in the present and into the future. The interacting influence of fire and climate varied by species, implying that the impacts of future shifts in climate and disturbances may alter forest composition, as species respond individualistically to each stressor. Likewise, the role of fire in mediating responses to climate will change through time as summer temperatures increase. These complex interactions will dictate the response of dry mixed-conifer forests to climate change and projected increased in fire extent in the future (Liu et al. 2010, Littell 2011).

Climate and fire interact to determine forest regeneration

Fire is an integral process driving regeneration of Douglas-fir. Fire moderates the regeneration environment under current climate conditions by increasing seedling recruitment where canopy opening occurs. We found fewer Douglas-fir seedlings where the density of live mature trees was greater than roughly 200 trees ha^{-1}, and where live seed trees were more than roughly 100 m away (Figure 3.2). Increased light availability and reduced competition (York et al. 2003, Gray et al. 2005, Moghaddas et al. 2008) likely improve the suitability of a site for Douglas-fir regeneration. However, in areas burned with high severity, high tree mortality contributes to lower seed source availability (Savage and Mast 2005, Haire and McGarigal 2010, Kemp et al. 2015a). Thus, tradeoffs between resource availability and seed source availability are important in determining regeneration patterns of heavy-seeded species such as Douglas-fir, which may disperse only fraction of their seeds more than 100 m from an edge (McCaughey et al. 1986, Bonnet et al. 2005). For example, Kemp et al. (2015a) found a threshold of approximately 75 m from a live seed source beyond which regeneration of Douglas-fir was unlikely on the same sites used in the current study.

Additionally, it is likely that seedling regeneration patterns are at least partially attributable to fire behavior, which in turn is a function of weather, fuels, and topography at the time of burning. This hypothesis is supported by our finding that the random effect of each individual fire event was significant in our models for both ponderosa pine and Douglas-fir, indicating that some of variability in seedling abundance can be explained by the properties of a particular fire. This finding underscores the importance of sampling multiple fire events in order to make broad-scale inferences.

Our findings that climate and fire interact to influence seedling regeneration have important implications when considering shifts in future fire regimes under a warming climate. Repeated fires or short intervals between fires may remove mature and regenerated trees that could have provided seed sources for subsequent tree regeneration (Keeley et al. 1999, Johnstone and Chapin 2006, Brown and Johnstone 2012). Likewise, if the relative size of high severity patches increases with future climate change, and more area experiences stand-replacing fires, the removal of seed sources could limit future regeneration (Kemp et al. 2015a). Although we did not quantify the impacts of increased fire extent and burn severity on seedling regeneration in the future, fire extent is projected to increase two to five fold by mid-century in the Northern Rockies (Littell 2011) which will lead to proportionally more area burned at high severity (Dillon et al. 2011). The interaction of increased area burned and more area of stand-replacing fire are likely to have important implications for future forest structure.

Climate also directly influences regeneration of Douglas-fir and ponderosa pine. We observed less regeneration of both Douglas-fir and ponderosa pine on sites with warmer mean summer temperatures. We evaluated seedling regeneration relative to mean climate conditions, yet, year-to-year variability and extremes in summer temperatures are also important for determining the success of post-fire regeneration (League & Veblen 2006). Summers in the Northern Rockies region are characteristically dry, with little consistent rainfall for several months, thus, the ability of seedlings to access soil moisture quickly may determine their success or failure during the first growing season (Daubenmire 1968).

Summer temperatures are therefore likely to not only drive patterns of the distribution of species over decades, but interannual variability will also be very important in the early stages of seedling establishment when seedling success is vulnerable to short-term (e.g., days to months) droughts and excessive heat (Kolb and Robberecht 1996).

Patterns of post-fire seedling abundance of Douglas-fir and ponderosa pine in our study region follow a trend of increasing abundance up to mid-elevations and decreasing abundance at high elevations (Figure 3.5). Higher elevation sites are projected to become more favorable for both ponderosa pine and Douglas-fir regeneration as the climate warms (Figure 3.5), suggesting a potential upslope shift in favorable sites for post-fire regeneration for these two species in the future. Temperature varies with elevation according to the environmental lapse rates, and thus mountainous regions with complex terrain have steep temperature gradients (Lookingbill and Urban 2003, Dobrowski et al. 2009). Fine-scale variation in climate, as modified by topography, is likely to drive patterns in species composition and abundance that will be observed in the future (Dobrowski 2011). For example, we may start to observe patterns of post-fire regeneration constrained to locations where topography buffers the effects of a warming climate. Dodson and Root (2013), for instance, observed a pattern of increasing post-fire regeneration success in dry mixed-conifer forests at higher elevations in eastern Oregon, suggesting that moisture stress at low-elevation, warm, dry sites was beginning to limit regeneration after disturbance.

We expected water balance variables, such as actual and potential evapotranspiration and water deficit, to be important, but they were not included in our final models. Topography is important for modifying solar insolation and microclimate; for example, north-facing aspects with less solar insolation were linked with greater tree regeneration success after fire for several conifer species on xeric sites in Colorado (Donnegan and Rebertus 1999). We suspect that our models may underrepresent the importance of these metrics for ponderosa pine and Douglas-fir. While temperature surfaces are driven by physical relationships with elevation, and often modeled at fine scales (e.g., 10-m digital elevation models), the spatial scale of AET, PET and other climate data (800-m) may have been insufficient to capture the variability in precipitation, soils, and energy at fine scales that contribute to the actual climate experienced by tree seedlings. Further, calculations of AET, PET, and water deficit use estimates of water demand from a standard reference crop, but seedlings such as ponderosa pine are likely to be more tolerant of lower soil water availability and desiccation than reference crops (Fowells and Kirk 1945).

The influence of climate and fire on regeneration will vary by species

Summer temperature was an important driver of both ponderosa pine and Douglas-fir abundance, but future temperature increases will differentially affect the two species. Thus, projected changes in the distribution of dry mixed-conifer forests following wildfire will vary by species. Our models indicated that Douglas-fir regeneration was unlikely above a mean summer temperature of 17 - 18° C, regardless of whether seed sources were available. By mid-century, the majority (82%) of sites that we sampled are projected to be above that temperature threshold for seedling survival, using the RCP 8.5 emissions scenario, leading to significant projected decreases in Douglas-fir regeneration on most (77%) of the sites we sampled. Ponderosa pine, on the other hand, has a higher temperature tolerance, and still persists above 17° C, implying that it is likely to persist at its current densities or increase on a greater proportion of sites.

The shift towards a warmer landscape in the future will favor more regeneration of ponderosa pine and Douglas-fir at mid- to high elevation sites. Currently, ponderosa pine occurs at a lower median elevation than Douglas-fir and may therefore replace Douglas-fir in some of the mid-elevation sites. However, because ponderosa pine regeneration is affected primarily by temperature, increases in temperature in the future are projected to drive more substantial increases in the elevation distribution where this species is likely to increase compared to Douglas-fir, which is predicted to respond to the interacting influences of temperature and fire (via distance to seed source). Given the significance of future summer temperatures to both Douglas-fir and ponderosa pine, however, low elevation sites currently occupied by dry mixed-conifer forests may have little regeneration potential post-fire in the future. Future fires in low-elevation forest thus have the potential to catalyze changes in species composition from dry forest to shrubland or grassland, especially if repeated fires result in high tree mortality coupled with poor regeneration (van Wagtendonk et al. 2012, Stevens-Rumann 2015)

The relative importance of fire and climate changes under future climate change scenarios

While the results from this study and others highlight the importance of wildfire in determining seeds sources for forest regeneration (e.g., Donato et al. 2009, Kemp et al. 2015a), our work further highlights that as temperatures warm, the relative importance of wildfire will decrease. For example, for Douglas-fir, once mean summer temperatures exceed approximately 17 -18° C, seedling regeneration will be poor regardless of seed sources. These interactions reflect the importance of varying stages of forest regeneration. Near-by seed sources are necessary, but not sufficient for successful germination. Likewise, seedlings may germinate but not survive for more than one to several years if the climate conditions are too warm. Although reduced tree density, presumably as a result of fire, was linked to higher Douglas-fir regeneration in the present study, our results suggest that regeneration in the future will be difficult with or without fire. Furthermore, the loss of canopy cover, which can moderate temperature extremes at the seedling scale, may amplify the impact of a warming climate on seedling regeneration, while retaining canopy cover could potentially help buffer future changes (Dobrowski et al. 2015). The combination of more mortality from fire and less recruitment because of less favorable climate conditions is likely to result in a future shift in the location of dry mixed-conifer forests on the landscape.

Implications for forests in a warming climate

The balance between mature tree survival and post-disturbance tree seedling recruitment will affect the long-term dynamics of forest composition and persistence (e.g., Johnstone et al. 2010). Predictions of trees species distribution shifts in the future rely in large part on understanding and identifying the niche characteristics of mature trees (Soberón and Nakamura 2009). Yet, species distribution changes will result, in large part, from the combined demographic effects of mortality and recruitment (Jackson et al. 2009). Where there is a mismatch between recruitment potential and conspecific adults, the regeneration niche will determine patterns of species occurrence, abundance, and recovery following a disturbance (Jackson et al. 2009, Dobrowski et al. 2015). In dry mixed-conifer forests of the northern Rockies, seedling regeneration of both Douglas-fir and ponderosa pine is currently most probable at locations that have a mean summer temperature between 14 - 16°C, though we sampled across the full range of sites that characterize the current occurrence of this cover type in the northern Rockies. This suggests that while mature individuals that survived the most recent fires may continue to persist in the future on many of these sites, the potential for post-fire regeneration is limited to a smaller subset of sites. As these locations become warmer in the future, the potential for regeneration will continue to be reduced. Where seedlings are currently established, the long-term impacts on the distribution and abundance of these species will depend on mortality at both the seedling and mature life stages.

Maintaining forest resilience is a high priority for forest managers with future climate change (Kemp et al. 2015b), and, thus, understanding where the landscape may or may not be able to support the regeneration and survival of valuable tree species will help forest managers prioritize re-vegetation treatments following disturbance or thinning. The establishment of trees in areas recovering from recent disturbances may help slow the projected impacts of climate change, as trees that establish under suitable conditions in the present may be less likely to succumb to warmer temperatures in the future. Over 40% of dry forests in the Northern Rockies have burned in the past century (Morgan et al. 2014) and these areas might be high priority for continued management through the use of low intensity prescribed fire or managed natural ignitions, where appropriate, to preserve seed sources and current species composition and to reduce the probability of future high severity fire that could potentially induce a shift in species composition or abundance.

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Table 3.1. Description of explanatory variables considered in generalized additive models predicting the abundance of seedlings post-fire. We required each model to have one variable from each category and that each of the explanatory variables be independent from one another.

Process	Category	Variable	Description
Climate*	Temperature	Summer temperature (°C)	Average June through September temperature
		Seasonality (°C)	Difference between maximum summer temperature and minimum winter temperature
	Water balance	AET (mm)	Annual actual evapotranspiration
		AET/PET	Ratio of annual actual evapotranspiration to annual potential evapotranspiration
		Water deficit (mm)	Annual water deficit, calculated as the annual potential evapotranspiration (PET) minus annual actual evapotranspiration (AET)
		Soil moisture (mm)	Mean summer (June - September) soil moisture
	Precipitation	Spring precipitation (mm)	Total March through May precipitation
		Summer precipitation (mm)	Total June through September precipitation
Fire	Canopy Cover	Canopy cover (%)	Canopy cover measured using a spherical densiometer; averaged from three locations on each transect
		Live tree stand density (trees ha^{-1})	Density of live trees measured using a 2 or 4 m basal area factor (BAF) prism post-fire; averaged from three locations on each transect
	Burn Severity	RdNBR	Landsat TM+ satellite derived Relativized differenced Normalized Burn Ratio (data from Dillon et al. 2011)
	Seed Source	Distance PSME (m)	Mean distance to the nearest live Douglas-fir seed trees, measured from the center of each transect
		Distance PIPO (m)	Mean distance to the nearest live ponderosa pine seed trees, measured from the center of each transect

* All climate variables have a 800 m resolution and were averaged for the 30 year period from 1981 - 2010

Table 3.2. Summary statistics of model fit for full and cross-validated negative binomial generalized additive models. We used a Chi-squared goodness-of-fit test to examine the fit of the negative binomial distribution to the distribution of our data, where the null hypothesis assumes the model and empirical data come from the same distribution. A failure to reject the null hypothesis indicates a good fit of the model to the data. Spearman's rank correlation was used to assess the correlation between the predicted and observed seedling densities. Cross-validation was accomplished through 1000 bootstrapped model runs using 75% of the data to train the model and the remaining 25% of the data for model validation. Models means and 95% confidence intervals are reported.

	Douglas - fir		**ponderosa pine**	
	Full Model	Cross Validation [95% CI]	Full Model	Cross Validation [95% CI]
Explained Deviance (%)	57.9	58.0 [52.0 - 63.4]	54.3	54.1 [45.3 - 62.2]
χ^2 GOF Statistic	1.02	1.02 [0.98 - 1.06]	0.56	0.56 [0.50 - 0.63]
χ^2 p-value	0.43	0.42 [0.30 - 0.55]	1.00	1.00 [1.00 - 1.00]
Spearman's ρ	0.81	0.76 [0.61 - 0.87]	0.50	0.11 [-0.15 - 0.34]

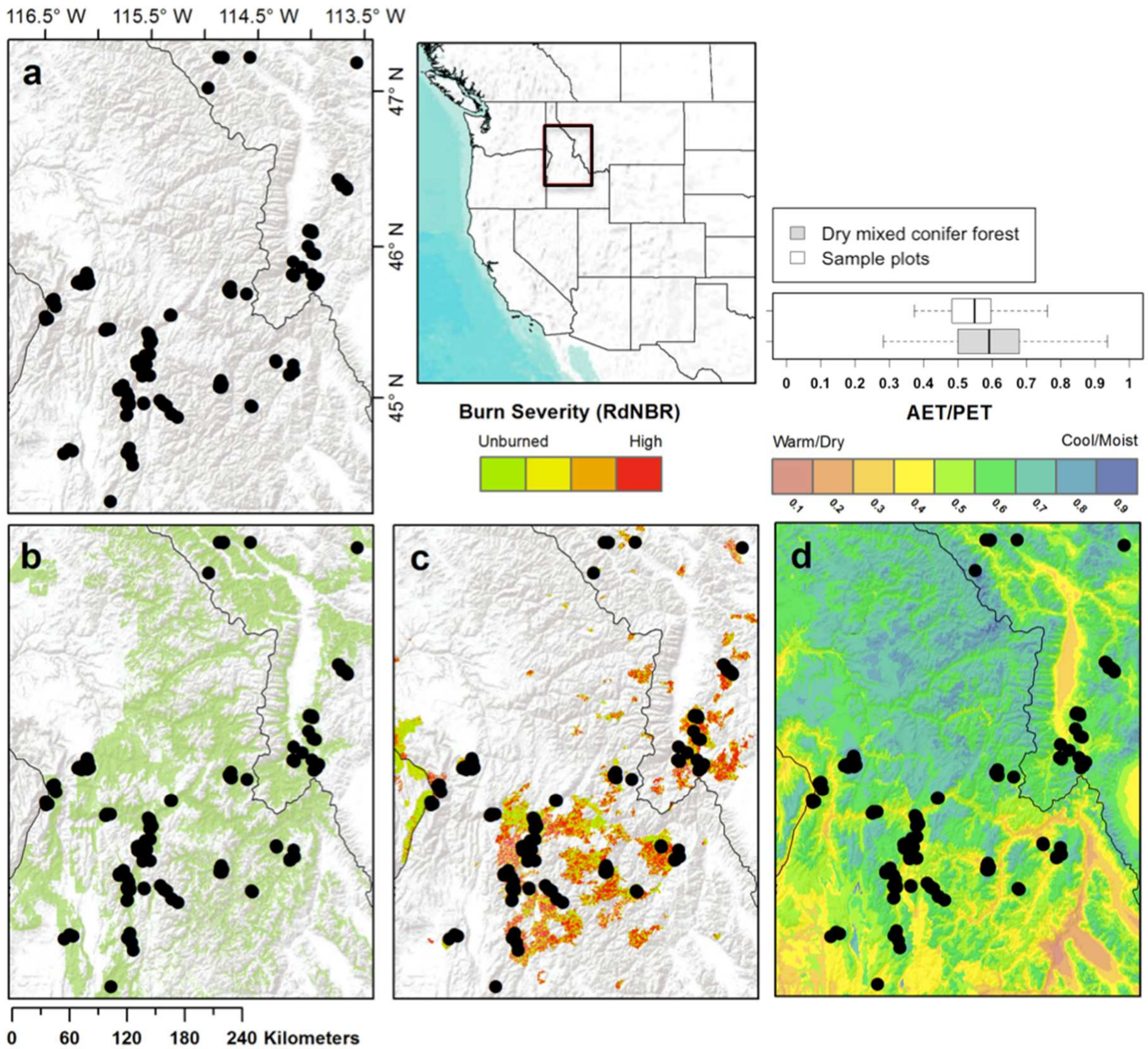

Figure 3.1. (a) We sampled 177 sites across central Idaho and northwestern Montana in complex mountainous terrain that fell within (b) dry mixed conifer forest type, (c) had burned in either 2000 or 2007 and that (d) represented a range of climate conditions characterizing the dry mixed conifer forest vegetation type (defined by the ratio of actual evapotranspiration to potential evapotranspiration). Our sites were slightly biased towards lower elevation warm dry sites across the range (boxplots). The ratio of actual evapotranspiration to potential evapotranspiration (AET/PET) ranges from 0 to 1, where warm dry sites have a lower AET/PET ratio than cool moist sites.

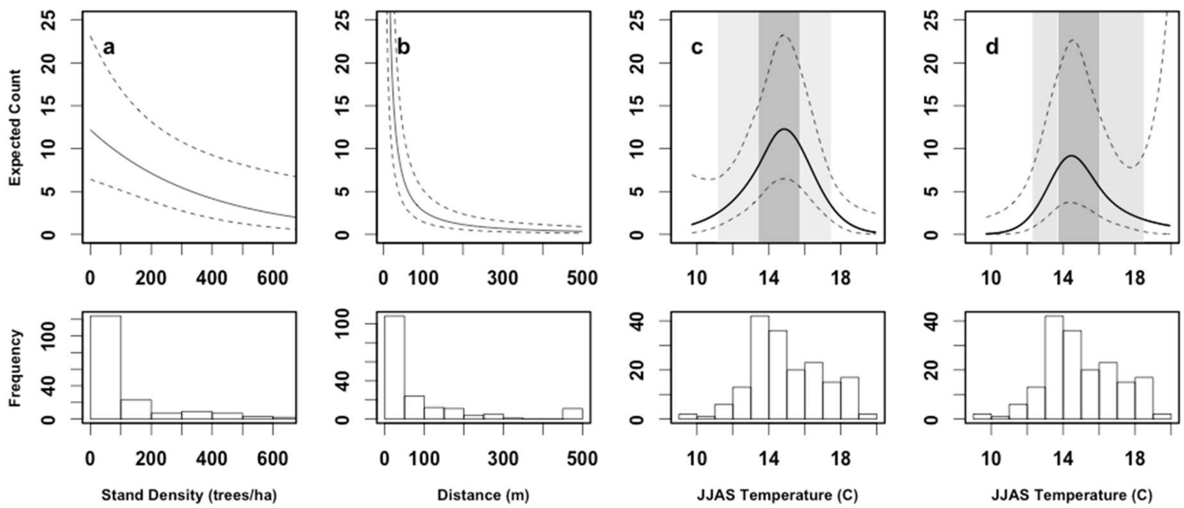

Figure 3.2. Partial dependence plots showing the effect of the explanatory variable on the expected counts of Douglas-fir and ponderosa pine seedlings while all other variables in the model are held constant at their medians. (a) Live tree stand density, (b) distance to a live Douglas-fir seed tree and (c) mean summer temperature were the most significant variables predicting Douglas-fir seedling abundance. (d) Mean summer temperature was the only variable significantly influencing ponderosa pine seedling abundance. In panels (c) and (d) the dark shaded area indicates the temperature range of 50% of the data under the fitted curve and the light shaded area 95% of the data. Dashed lines represent the 95% confidence intervals around the response and are wider where there are fewer samples along the data distribution. Histograms display the distribution of site data used to develop the generalized additive models.

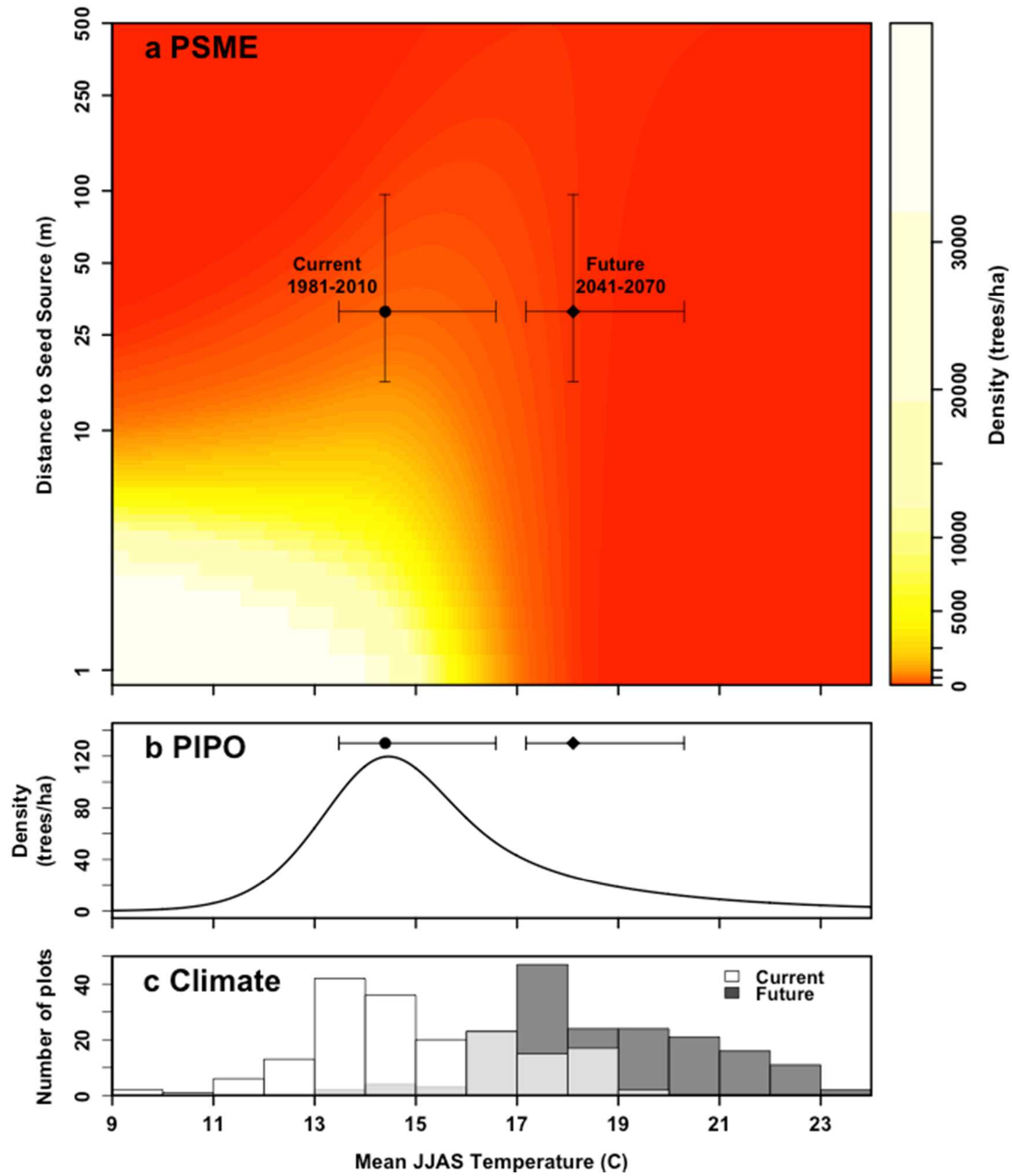

Figure 3.3. Contour plot showing predicted (a) Douglas-fir seedling densities for a range of different temperatures and distances to seed source trees. Points represent the current median summer temperature and distance to seed source and the future predicted seedling densities based on CMIP5 RCP 8.5 model mean projection of summer temperature change at each of our sites for the period 2040 – 2070. Whiskers extend to encompass the 25th and 75th percentiles of the data (b) Predicted ponderosa pine seedling densities for a range of different temperatures with the innerquartile range of current temperature and future temperature represented by the bars, and (c) the number of plots we sampled in each current (white) and future (dark grey) temperature bin. Overlap between current and future temperatures is displayed by the light grey bars.

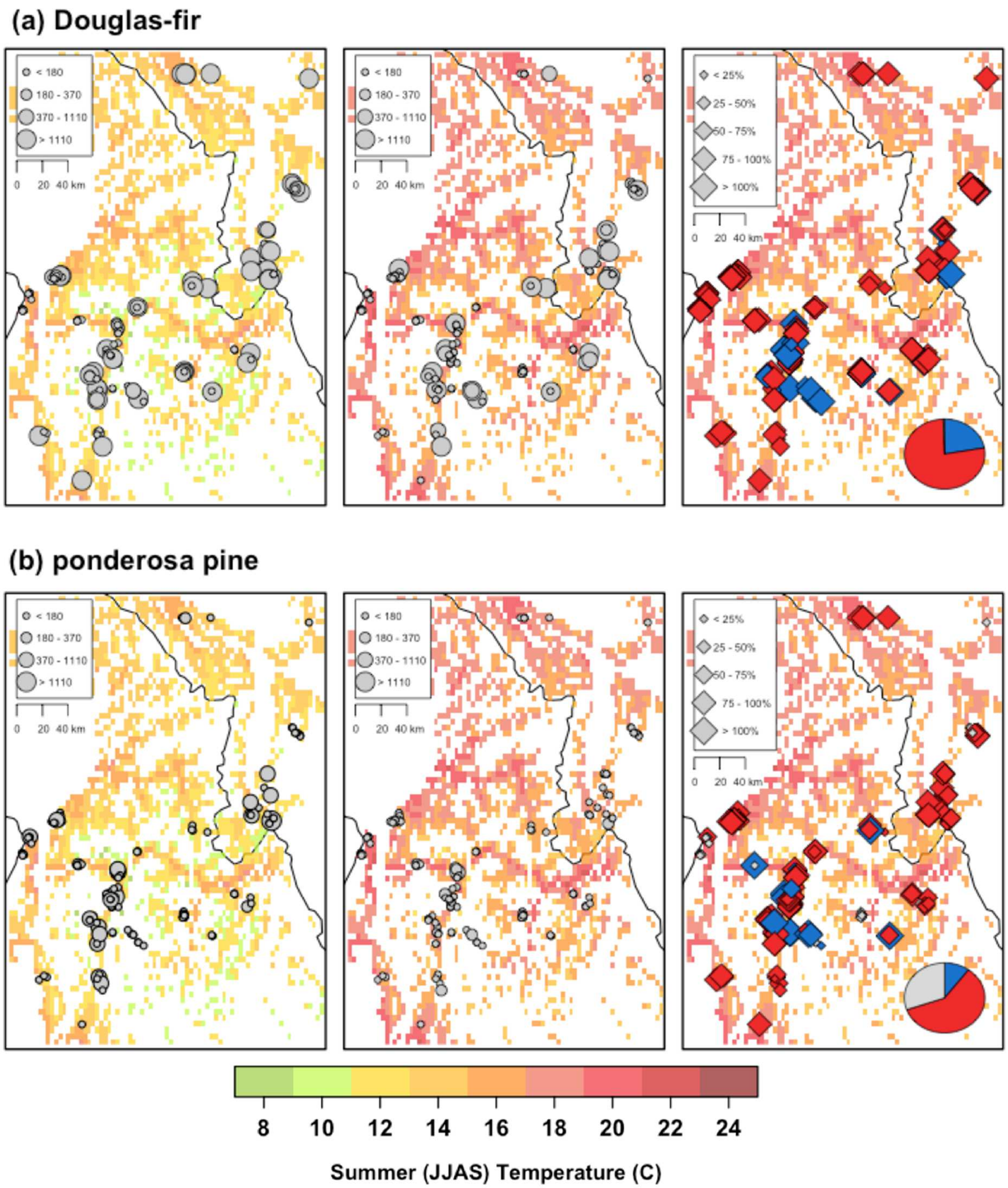

Figure 3.4. Map of current (left), predicted future (center), and change (right) in (a) ponderosa pine and (b) Douglas-fir seedling densities. The background is the average summer temperature of dry mixed-conifer forest currently (1981-2010) and as predicted for mid-century (2041-2070) under the RCP 8.5 scenario. Seedling densities are considered to have changed if the density increased or decreased by at least 1% over the predicted period. Pie charts display the percent of sites with increase (blue), decrease (red) and no change (grey) for each species.

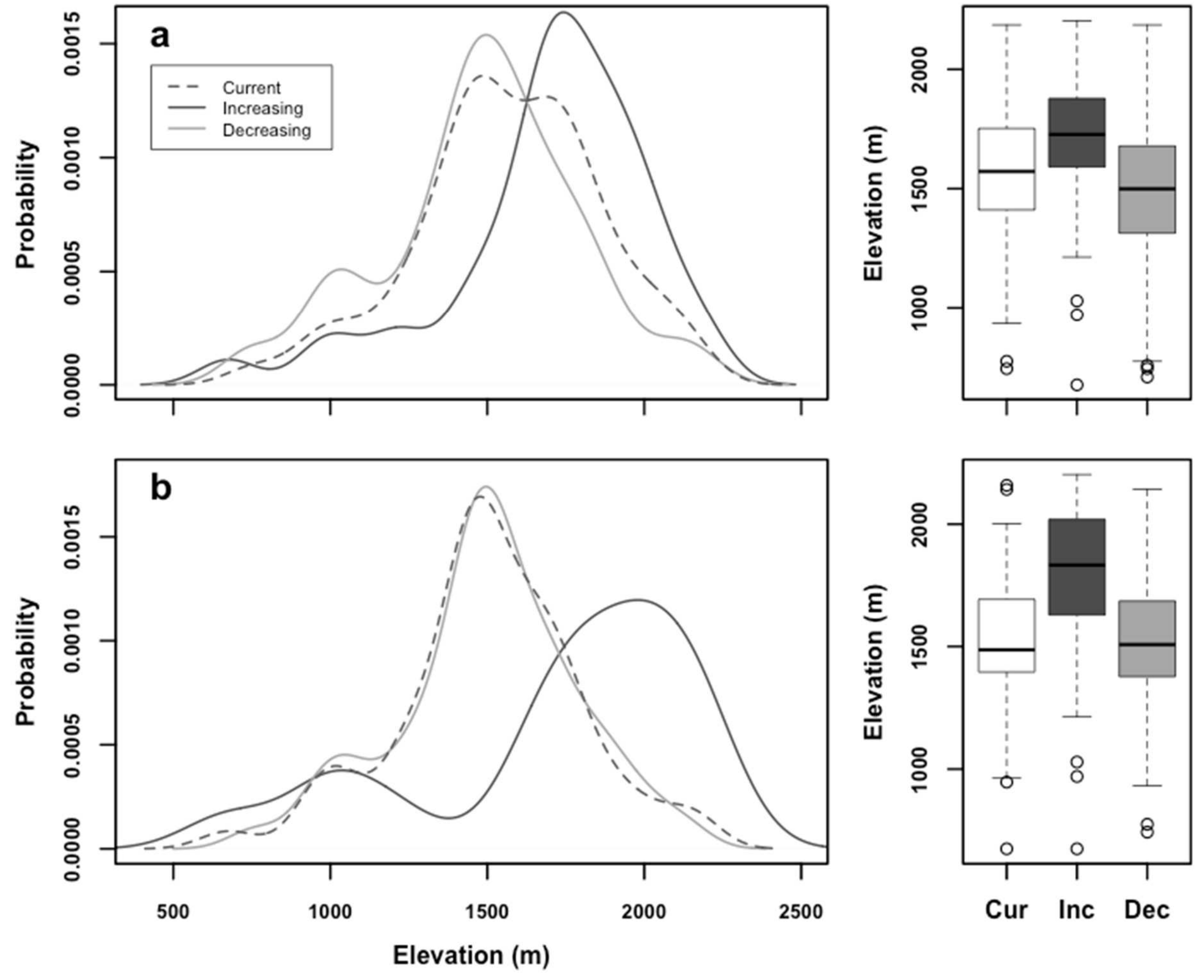

Figure 3.5. Probability density functions for the elevation of sites where seedling densities are predicted to increase or decrease relative to current modeled densities for (a) Douglas-fir and (b) ponderosa pine. Sites were defined as having increasing (decreasing) densities if there was a positive (negative) percentage change between the predicted current and predicted future seedling densities. Boxplots show the 25^{th}, 50^{th} and 75^{th} percentile and whiskers the 10^{th} and 90^{th} percentile elevation of sites where seedlings of each species are currently found, predicted to increase, and predicted to decrease in the future. Circles represent sites outside of the central 90% of the data.

CHAPTER 4

Managing for Climate Change on Federal Lands of the Western U.S.: Perceived Usefulness of Climate Science, Effectiveness of Adaptation Strategies, and Barriers to Implementation

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Abstract

Recent mandates in the United States require federal agencies to incorporate climate change science into land management planning efforts. These mandates target possible adaptation and mitigation strategies. Yet, the degree to which climate change is actively being considered in agency planning and management decisions is largely unknown. We explored the usefulness of climate change science for federal resource managers, focusing on the efficacy of potential adaptation strategies and barriers limiting the use of climate change science in adaptation efforts. Our study was conducted in the northern Rocky Mountains region of the western U.S., where we interacted with 77 U.S. Forest Service and Bureau of

Land Management personnel through surveys, semi-structured interviews, and four collaborative workshops at locations across Idaho and Montana. We used a mixed-methods approach to evaluate managers' perceptions about adapting to and mitigating for climate change. Although resource managers incorporate general language about climate change in regional and landscape-level planning documents, they are currently not planning on-the-ground adaptation or mitigation projects. However, managers felt that their organizations were most likely to adapt to climate change through use of existing management strategies that are already widely implemented for other non-climate related management goals. These existing strategies (e.g., thinning, prescribed burning) are perceived as more feasible than new climate-specific methods (e.g., assisted migration) because they already have public and agency support, accomplish multiple goals, and require less anticipation of the future timing and probability of climate change impacts. Participants reported that the most common barriers for using climate change information included a lack of management-relevant climate change science, inconsistent agency guidance, and time and resources needed to access, interpret, and apply current climate science information to management plans.

Keywords climate change • adaptation • land management • public lands • decision making • Forest Service • Bureau of Land Management

Introduction

In the U.S. northern Rocky Mountain region, the United States Forest Service (USFS) and Bureau of Land Management (BLM) are responsible for managing public lands that account for roughly 13 million hectares in Idaho and 11 million hectares in Montana (Gorte et al. 2012). Climate change is likely to impact the forests and rangelands managed by these agencies and alter important ecosystem services, such as fresh drinking water sources, recreation, and timber production, all of which are integral to local communities and economies (e.g., Pederson et al. 2006). Therefore, how these agencies adjust their current management practices to adapt to and mitigate the impacts of climate change will be an important aspect of future land management.

Federal agencies have emphasized climate change within planning and management at the national level for several years. In 2008, the USFS acknowledged the role climate change has played in changing wildfire regimes, bark beetle infestations, and water availability, stating that, "Without fully integrating consideration of climate change impacts into planning and actions, the Forest Service can no longer fulfill its mission" (Dillard et al. 2008, pg. 2). The USFS highlighted two strategies for addressing climate change impacts on national forests: facilitated adaptation, i.e., actions for reducing the negative impacts of climate change; and mitigation, i.e., actions to reduce emissions and enhance natural carbon sequestration (Dillard et al. 2008, Cruce and Holsinger 2010). Additionally, the USFS created the "Climate Change Performance Scorecard," which was intended to help units within the agency implement "short-term initiatives" and "long-term investments" in response to projected impacts of climate change, as well as track their progress towards these goals (Tkacz et al. 2010). Likewise, the BLM has had a strategy for responding to climate change in place since 2001, though potentially less targeted than the guidance put forth by the USFS (Ellenwood et al. 2012). Furthermore, as part of Secretarial Order No. 3289, in 2009 the Department of the Interior (DOI) established several Climate Science Centers and Landscape Conservation Cooperatives to address informational concerns and anticipated challenges the DOI may face in managing for the impacts of climate change (GAO 2007). Presidential executive orders issued in 2009 (EO 13514) and 2013 (EO 13653) also provided uniform policy guidance aimed at encouraging climate change adaptation and carbon mitigation within all federal agencies.

Although federal mandates are in place, addressing climate change at regional (unit/forest/watershed) and local (field office/district/stand) levels presents numerous challenges, especially within impact assessments designed for long-term land use planning or specific management projects. These challenges include both internal and external factors, such as a lack of agency direction (Archie et al. 2012), time and funding allocated for implementing new programs, litigation by external interest groups (Lachapelle et al. 2003, Jantarasami et al. 2010, Wright 2010), or negative public perceptions (Archie et al. 2012, Archie 2013). Transferring science between research and management can also be a challenge. Managers often lack time to review relevant literature (Kocher et al. 2012) and a dearth of information at management-relevant scales can impede the use of existing science (Archie et al. 2012). For example, resource managers have repeatedly expressed a need for downscaled climate change projections to match the scales at which land management is accomplished (e.g., Jantarasami et al. 2010, Archie et al. 2012).

Climate change may also dictate that land managers consider novel approaches to land management to achieve their goals. Rangeland and forest management in the western U.S. often emphasizes evaluating current conditions against historical reference conditions and using the estimates of the degree of ecosystem change to prioritize different types of treatments (Keane et al. 2009, Caudle et al. 2013). The extent of change from past decades and centuries, coupled with predicted future changes, suggests that adaptive management approaches that consider a wide range of different options may be necessary to effectively carry out the provisions highlighted in agency policies for climate change (Hobbs et al. 2014).

Although climate change has been highlighted as an important management priority at the federal level, it is still uncertain how climate change science is being considered in project management and planning by local resource managers. Our research addresses how federal land management agencies in the U.S. northern Rocky Mountains are currently utilizing or thinking about applying climate change science to management activities. Specifically, we asked USFS and BLM managers how climate change science is useful for their work and whether they, as individuals, or their agencies are currently incorporating this information into land management planning. Additionally, we asked what management actions they see as effective for adapting to, or mitigating, climate change and if their agencies are considering implementation of these actions. Finally, if managers are not addressing climate change in their planning efforts as suggested by the policy directives, what barriers do they perceive impede their use and incorporation of it into management? Understanding the challenges resource managers perceive and the techniques they are using to adapt to the impacts of climate change will help to highlight the types of information, policies, and directives that can better aid managers in incorporating climate change science into management.

Methods

We used a series of different approaches, including quantitative surveys, semi-structured interviews, and one-day workshops, to understand managers' perceptions about the usefulness of climate change science, efficacy of potential adaptation strategies, and barriers to implementation of adaptation and mitigation measures. Survey and interview input was collected from study participants both before and after the workshops as part of a larger study to track individual changes in perceptions about climate change science (Blades 2013). In this article, we aggregate individual responses from the surveys, interviews, and workshop discussions to focus on general tendencies and insights across participants, drawing on pre- and post-workshop responses that bear on our research questions where appropriate, rather than analyzing individual changes from pre- to post-workshop.

The vast majority (> 90%) of federal lands in Idaho and Montana are managed by the USFS and BLM, and these lands account for approximately 62 and 29%, respectively, of the land base of these two states (Gorte et al. 2012). Therefore, we elected to focus the majority of our recruitment efforts on these two agencies (USFS & BLM), though other federal (e.g., USFWS, NPS, NOAA), tribal, and state resource managers were invited to participate in our workshops. Participants who were likely to actively make or implement land management decisions, and whose agency experience would give them the ability to comment in depth on land management and climate change directives were purposively selected through public contact lists for the study. These participants were planners, ecologists and biologists, silviculturists, fire managers, and water resources managers. After the initial selection of participants, a snowball sampling approach was employed where individuals who agreed to participate were asked to identify other interested individuals or co-workers who would have knowledge of how agencies address climate change. We initially recruited 257 individuals to participate in the workshops, however; only 97 individuals elected to participate (38% response rate). Of those 97 individuals, 77 were federal land managers from the USFS (n =

66) and BLM (n = 11). All participants that signed up for the workshops were sent pre-workshop surveys, and a random sample of those responding to the surveys were asked for interviews. We elected to exclude individuals from other state and federal agencies because of the overall poor response rate from these agencies. We aggregated USFS and BLM responses for all phases of data collection based on the small representation by BLM employees. Both the USFS and BLM are mandated to manage for multiple uses and sustained-yields, and though these specific uses may differ slightly (e.g., timber harvest vs. cattle grazing and mining), many are similar (e.g., recreation, wildlife, water; Gorte et al. 2012). Likewise, both agencies must allow public participation in the planning process and address potential environmental impacts as part of the National Environmental Policy Act (NEPA) of 1969.

We conducted our one-day workshops in four locations across the northern Rocky Mountains in November 2012 (Figure 4.1). The locations represented five national forests and two BLM districts (Figure 4.1). Representatives of several collaborative organizations and non-profit groups who actively work with individuals from the USFS and BLM participated in the workshops, but here we focus specifically on federal resource managers' responses. During each 8-hr workshop, we presented historical information and future projections about climate change impacts at global, regional, and local scales. Most of the regional and local scale projections focused on changes in the northern Rocky Mountain region for resources of interest, including hydrology, forest species distributions, and wildfire activity. At the end of each presentation, workshop participants were assigned to small groups chosen to represent the mix of agencies, organizations, and specializations present. During these breakout discussions, participants were asked to reflect upon how climate change information could be useful in their work and important management implications of the information presented. Discussions were facilitated in a manner to provide all participants an opportunity to speak openly about their personal perceptions as well as to express opinions on behalf of their agencies. Main discussion points that arose during the conversations were recorded on flip-charts by trained facilitators.

Online surveys were sent to all workshop participants via e-mail prior to and immediately following the workshops. If participants had not completed an online survey prior to arriving at the workshop, they were asked to fill out a paper copy upon arrival. Quantitative survey responses were made on a 7-point Likert-type scale from -3 (strongly disagree) to + 3 (strongly agree) and descriptive statistics were summarized using SPSS version 13 (SPSS 2010). Where survey questions were asked both before and after the workshops, we present the results of only the pre-workshop data because we believe these data most closely reflect perceptions of the broader population of resource managers who have not participated in workshop presentations, discussions, or conversations. Several of the post-workshop survey questions explicitly asked participants about the perceived usefulness of information at varying spatial scales (global, regional, and local); we compare these ratings using one-way analysis of variance.

Pre-workshop interviews were conducted by phone in late October and early November, 2012. Questions during the pre-workshop interviews covered a range of topics related to perceptions of climate change and impacts, including credibility and salience of climate change science, perceived vulnerability to and severity of climate change impacts, and individual and collective management responses (Blades 2013). However, for the purposes of this analysis, we only included responses regarding the current use of climate change science, potential actions for adapting to and mitigating the effects of climate change, and barriers to using climate change science within management organizations (Appendix H). All interview questions were open-ended, allowing for a range of responses, and interviewers asked follow-up questions to clarify responses. Post-workshop interviews were conducted by phone in December 2012 and January 2013 and generally covered the same topics as the pre-workshop interviews, but also included evaluative components targeted to give the researchers feedback on the workshop materials and process (data not reported here; see Blades 2013).

Phone interviews were digitally recorded and transcribed. Following transcription, initial codes (high-level themes) were developed by one researcher and then evaluated by other members of the research team for clarity and completeness. The same codes were used for the pre- and post-workshop interviews and discussion group themes. After several initial rounds to refine the coding rules, all interviews and discussion points were coded using NVIVO 10.0 software (NVIVO 2012) by one researcher. A subset of interviews and discussion points was subsequently coded by a second researcher to establish reliability (kappa = 0.80; Krippendorff 2004). Sub-themes were subsequently developed under each high-level theme (code) using a peer-debriefing process where each researcher independently established important and cross-cutting points from the interviewees and the group summarized and corroborated common themes (Appendix K).

Results

We interviewed 60 individuals prior to participating in the workshops; 35 of those individuals were also interviewed after the workshop. In all, 77 resource managers participated in the four workshops, 61 of whom completed both the pre- and post-workshop online surveys. The responses we received from repeatedly engaging participants through different mediums allowed us to sufficiently understand managers' perceptions and to reach saturation of themes during the interviews and breakout discussions (Bowen 2008). In presenting results below, we integrate excerpts from interviews and workshop discussions chosen to exemplify the general themes we distilled from across the data sources we collected (Bansal and Corley 2012, Poortman and Schildkamp 2012).

Usefulness of climate change science

The majority of survey participants thought climate change science was useful for their work (90%), for future planning efforts (97%), or for specific management projects (80%; Figure 4.2). Furthermore, more than 80% of the land managers surveyed agreed or strongly agreed that using climate change science was within their job description or responsibilities (Figure 4.2), indicating an awareness of national policies aimed at adapting to and mitigating climate change. When asked in interviews and workshop discussions how climate change science is currently being used, many participants mentioned that it is addressed in environmental impact statements (EIS), environmental assessments (EA) and forest plans that have been recently revised, along with other disturbance factors (e.g., wildfire, bark beetles, floods). However, these documents often contain only broad, non-specific language. For example, one hydrologist mentioned that "cursory statements are put into our EISs or EAs, and it's more like checking a box than it is really looking into what… could be the potential effects [of climate change]."

During the workshop discussions, participants emphasized that climate change projections were useful for showing that adaptation may be necessary, but less useful in understanding how to adapt. This uncertainty about the best adaptation strategies meant that many of the resource managers we interviewed were unlikely to change their management practices to accommodate future change. For example, a timber manager from the Forest Service noted that he was not going to “change the species compositions when I prescribe a plant in a re-vegetation harvest area.” Rather, he emphasized he would use the “stand dynamics [of] what has been there” to influence his “decision on what we're going to [plant in that stand] in the future.” Likewise, managers found it difficult to understand how to incorporate climate change science into their planning efforts. For example, one planner noted that many of the Forest Service’s management actions are still based on “our current understanding of climate being relatively static.” This planner went on to emphasize that “We’re not sure [of] the extent of climate change or what a 3°C increase in the global [mean temperature] means to us here locally. That’s the problem, we know that there’s a change globally, but what does that mean here on our 250,000 acres that we manage in northwest Montana? That has yet to be defined for us at a level we can [base] management decisions on.” Like this individual, many other participants pointed out that “project level planning [takes place over] pretty short time periods (5-10 years)” and at the scale of hundreds of acres, requiring “very site-specific analysis,” whereas climate change occurs over long periods and specific local impacts are difficult to predict. Thus, the current global and national-scale climate change projections are not very applicable for planning on-the-ground management activities.

Of the three spatial scales of information presented during the workshops (global, regional, and local), regional and local climate change projections were considered more useful for land management than global projections ($F_{3, 234} = 11.87$, $p << 0.001$; Table 4.1).

However, interviews and workshop discussions revealed a more nuanced interpretation of the usefulness of different scales of information. Discussions during the workshop revealed that participants felt that "local-scale models lacked site-specific data" or that "there was too much variability" at this scale. One silviculturist felt that local-scale models had to consider "so many variables and so many complexities in the natural system" and that modeling those types of processes was "really hard." Workshop participants did comment that, conceptually, the scale of regional projections was useful for thinking about "potential consequences or priorities" and "desired future conditions" across the broader landscape.

Management to address the impacts of climate change

Participants were asked during the interviews and workshops if there were specific actions they felt would be useful for adapting to and mitigating the effects of climate change on federally managed public lands in Idaho and Montana. Surveys addressed 10 specific management strategies that could be implemented to adapt to climate change; participants were asked to evaluate the likelihood and effectiveness of each of these strategies (Figure 4.3). Actions considered most effective were forest treatments to reduce fire hazard and improve forest health, such as thinning projects aimed at decreasing tree density or removing hazardous fuels, and infrastructure modification, such as replacement of existing roads and culverts to make them less flood prone (Figure 4.3). For example, one interview participant noted that "Upsizing culverts to prepare for earlier spring [snow] melts, or more precipitation falling as rain during that time period where it might be snow instead" could be effective for adapting to climate change. Participants also felt that infrastructure modifications, forest treatments to improve forest health and reduce fire hazard, and prescribed burning were the management actions that were most likely to be carried out by the USFS or BLM in response to potential climate change impacts. Restoration using alternative tree species or varieties that might be more resilient to climate change was considered potentially effective by participants, but less likely to be used by their agencies (Figure 4.3). For example, one manager commented, "we have not gotten into the mode of assisted migration or changing our species that we're planting because of what we think may happen in the future as the climate changes." Finally, participants felt that actions such as forest thinning to increase water availability (e.g., targeted thinning of conifers encroaching into wet meadows or semi-arid shrublands) or the intentional movement of species to areas or habitats predicted to be favorable in the future but currently outside their range (i.e., assisted species migration) were neither likely nor effective (Figure 4.3).

Although a few participants mentioned specific adaptation strategies during the interviews and breakout discussions, most participants felt uncertain about potential management actions that could help their agencies adapt to or mitigate climate change. "I think we are challenged to sort out what [to do] about climate change... we don't really know what we can do... I think we all realize that we are sort of bystanders to this," said one participant. Consistent with the surveys, participants who discussed specific management treatments for adapting to climate change in their interviews focused on using familiar techniques (e.g., thinning, prescribed burning).

Nearly half (46%) of the interviewees emphasized increasing "resilience" of forests for multiple objectives in their comments about how climate change adaptation might occur. Increasing resilience was also a common theme of group discussions during the workshops. For example, a planner with the BLM mentioned that the agency is "trying to make sure our streams are as resilient as possible—[so we do a lot of restoration activities to] remove the stream barriers, fish barriers, things that would warm temperatures...” Another planner mentioned that because climate change is “an uncertainty that we can’t necessarily predict and/or manage for,” the best management option might be to manage for a diversity of “[tree] age classes and species” to have something that might be “resilient in the future.” Resilience has been emphasized in many of the federal climate change policies, and this concept seemed to resonate with resource managers’ thinking about potential adaptation strategies.

Several participants expressed frustration that the amount of land they could effectively treat would be minimal compared to the potential impacts of climate change. “I’m looking at a map right now... and I’m [thinking] I could do something on the ground that would cost a bunch of money [and it] would be great, but in the grand scheme of things, it would only be a tiny, tiny piece of ground that I’m actually doing any good on,” commented one ecologist. Participants also recognized that the scale of land management being done currently might not be effective in mitigating climate change (i.e., reducing carbon emissions). For example, one forester from a regional USFS office noted that because of the extensive vegetated area his agency manages, “there are carbon storage issues that we could deal with in terms of reducing fire hazard and the large mass of carbon released from wildfire events.” However, this forester went on to comment that social barriers (e.g., litigation by environmental interest groups) limit the amount of area they can effectively treat. Because of these limitations, resource managers felt that they would instead be forced to adapt their management to deal with the impacts of climate change after the fact. “Our projects aren’t going to affect [climate change] but we will be affected by it, so what is our management going to do to respond?” one participant asked. “Adaptation is probably going to be the key,” noted another.

Furthermore, although previous policy has guided land management to consider historical reference conditions as a baseline for restoration, a few of the interviewees recognized that, in light of climate change, restoring to those conditions might no longer be a viable goal. For example, one silviculturist stated that "the thinking [in the USFS]... has been that if we restore things to within the historical range of variability, we somehow increase resistance and resilience to change. Now, we have to construct what could be the [future] ranges that will function with climate change." Seventy percent of participants surveyed prior to the workshop agreed that their agencies might be willing to explore alternative management solutions beyond restoring to reference conditions. Nevertheless, the totality of the survey, interview, and workshop data suggest that managers don't know what those solutions should be.

Barriers to use and implementation of climate change science

Our survey included three potential barriers based on the literature: time, funding, and politics (Figure 4.2). Time was mentioned frequently in the workshop group discussions; representative comments included "there is not enough time to learn new tools," and "there are so many other priorities, [that] climate change is just one more thing [that requires] time." Participants also commented that part of the difficulty in using climate change science was keeping up with the wealth of information that is continuously being published, where there is little time to "know all the latest, greatest science that's out there, and to have it readily available at your fingertips." Being able to readily access information in a concise format would reduce some of the perceived barriers participants had with using the science. Participants also elaborated on the issue of funding for climate change adaptation projects. For example, on regional planner emphasized that without "extra resources in terms of capacity or funding, how are [resource managers] supposed to do [anything about climate change]?"

External politics and litigation by public interest groups also appeared in the interviews as a major barrier that participants perceived to limit their ability to manage for the impacts of climate change in the future. Managers noted that much of their energy was devoted to dealing with issues that were of current concern to the public, leaving little time to focus on new issues like climate change. "You can [only] do so many projects and so you don't spend a lot of time on things that you're not being challenged on. The climate change [issue] seems to be an emerging issue that we're not actually pursued on yet. The things that you get pursued on are the ones you start paying more attention to," one forest planner noted in an interview. Another planner commented that even though "the Washington office [of the Forest Service says] we're [going to do] more accelerated restoration, and massive thinning [to mitigate for climate change], the reality is we get appealed and then we get litigated." This planner went on to say that managers "can't do anything on the ground without getting through the [environmental] issues, [which is] really such a sociopolitical piece of [management]."

Beyond the items included in the survey, participants discussed several other institutional barriers in workshop discussions and interviews related to using climate change science for management decisions. For example, the size of the agencies and the associated bureaucracy means that changes occur slowly and new ideas and management strategies are unlikely to catch on quickly. Comments from USFS employees such as, "the Forest Service is a big machine... with a lot of ingrained ideas of what we do," or, "because of the bureaucracy, things happen very slowly," were widespread throughout the group discussions and interviews. Many participants commented that in recent decades agencies have often operated reactively, dealing with issues after they become problems rather than anticipating situations and proactively addressing them. For example, one hydrologist remarked, “It’s not like we are waiting for [climate change] to come in. It’s been here for a couple of decades. We haven’t changed things, really. We’re talking about how we’re going to do this, and we should be talking about how we should have done this.” Although slow, some participants were optimistic that changes in ingrained management practices would eventually occur. One participant gave this example, “To change livestock grazing, for example, [might be] kind of a hard thing to do, but it seems like when people aren’t meeting permit stipulations that things will have to change. It might take a while before they realize actually that this is not just a weird year, this is a weird decade, [and] we are still not meeting targets year after year.”

Additionally, several participants noted a lack of organizational capacity to address climate change; that is, people are not trained and/or educated about climate change and there is no time or funding to support this effort, and, even if managers have the training, the expertise, or the inclination, the support and direction from higher levels may be lacking. One forest planner acknowledged that “[Climate change] is a stated policy of the Forest Service, there’s no question about that. But that doesn’t mean every district ranger, every forest supervisor, believes in it. That then gets reflected in their program of work and what they emphasize.” Another manager relayed a similar view: “We have the people, we have the experience and expertise and technical savvy to get this done. We need the support to be able to do it.” Participants emphasized that upper-level decision makers (e.g., district rangers, forest supervisors) had the final say on what projects get done on national forests and rangelands, therefore, it was “up to [the decision makers] to decide whether they want to take a risk or not [to do something about climate change].” Poor organizational support meant that these managers had little motivation to incorporate climate change into their planning efforts unless they were getting specific direction from these line officers.

Discussion

Many of the federal resource managers we interacted with from Idaho and western Montana USFS and BLM offices think that climate change science could be useful for the work they do, demonstrating that they consider climate change to be a salient issue with the potential to impact the resources they manage. Except for brief and oblique mentions of “climate change resilience” in land-use plans, few of the public land managers we surveyed indicated that they were actively using climate change science in their work. This result is likely to vary depending on the specific district or forest that individuals work on; for example, other national forests and BLM districts, such as the Okanogan-Wenatchee, Colville, and Olympic National Forests, in the nearby state of Washington, have been proactive about incorporating climate change science into forest-wide strategic plans (West et al. 2009, Halofsky et al. 2011) and are likely to have much more comprehensive guidelines in place for addressing the impacts of climate change in their planning efforts. However, our findings were consistent with results from interviews with natural resource managers in the southern Rocky Mountains about the usefulness of climate change science (Ellenwood et al. 2012), suggesting that the integration of climate change science into management planning may still be evolving.

The usability of climate change science is influenced by whether an appropriate scale of information exists and if the science is informative within the specific end-user decision making context (Dilling and Lemos 2011). Our results indicate that science at temporal and spatial scales that matched the scale of project planning was an important consideration when participants were evaluating the usefulness of climate change science. In our workshops, local- to regional-scale information that emphasized risk management and long-term planning, such as watershed projections of changing precipitation phases (Klos et al. 2014), monthly streamflow, and flood risk (Hamlet et al. 2010), were considered especially useful by resource managers. Downscaled climate change projections that focus at sub-regional scales and project impacts over shorter time frames are likely to be much more applicable to managers' goals (Letson et al. 2001, Rayner et al. 2005, Dilling and Lemos 2011, Archie et al. 2014). Where these resources can be made available through freely available outlets such as websites or personal blogs, they are more likely to be successfully accessed and applied to project planning (Archie et al. 2014).

Science that is "co-produced" between managers and scientists and tailored for specific resources or targets potential actions has also been shown to effective in overcoming informational barriers (Lemos and Morehouse 2005, Joyce et al. 2009, Dilling and Lemos 2011, Kocher et al. 2012, Littell et al. 2012, Moss et al. 2013). This approach has been used effectively in wildland fire (Kocher et al. 2012) and water resources management (White et al. 2008, Wilder et al. 2010). For example, hydrologic studies indicating the quantity or timing of available water sources can dictate how water is allocated for agriculture, development, or other uses (e.g., White et al. 2008). In its application to wildland fire management, forecasts informed by current science are used to allocate appropriate resources for the coming fire season. Science that focuses on management-relevant objectives and needs, such as information on fuels and long-term weather forecasts, is used to make decisions in the face of uncertain potential outcomes (Lemos and Morehouse 2005). Organizational structures that help bridge the boundary between science and management (e.g., boundary organizations) are likely to be key in maintaining an environment where scientists and managers can continually discuss relevant needs (e.g., White et al. 2008). In some cases, these structures already exist in the U.S. northern Rocky Mountains, where USFS funded Collaborative Forest Landscape Restoration Projects encourage collaborative, science-based ecological restoration. Though the goal of these organizations is not climate change adaptation, per se, as these organizations become institutionalized, they could serve as effective vehicles for knowledge production and sharing across administrative boundaries (Gaines et al. 2012). Approaches such as single or multi-day workshops or focus groups may also be effective for helping managers develop general adaptation strategies to deal with climate change (Littell et al. 2012, Blades 2013).

While climate information at management-relevant scales is starting to become available in the research community, access to that information may still be an issue for managers looking to use this information (e.g., White et al. 2008). Where information is accessible, it is often in a format that is difficult for managers to digest and apply. Several participants stated that they had neither the time nor the expertise to sort through the climate change science that is currently available. National forests have attempted to bridge this gap by creating regional and forest-specific climate change coordinator positions (Tribbia and Moser 2008). These individuals are responsible for collaborating with scientists to create, compile, and disperse regional climate change science relevant for each forest. However, the degree to which this task is effectively carried out often depends on the individuals, their motivation, and their other job responsibilities. This variability was evident in our study; participants in one workshop location were well informed about climate change projections and impacts due to effective communication between their regional climate change coordinator and personnel at the forest and district levels. However, participants in other workshops were not nearly so well informed. Prior studies have emphasized the importance of colleagues as information sources for federal resource managers (Tkacz et al. 2010), thus, well-informed climate change coordinators and line officers may play an essential role in getting climate change science incorporated into day-to-day land management activities (Archie et al. 2014).

In addition to a lack of management-relevant information, specific agency guidance, lack of resources (e.g., time, funding), and public support were the most frequently mentioned constraints when our study participants were asked to elaborate on barriers that prevented their use of climate change science. Although we did not separate responses from BLM or USFS participants, prior studies indicate that these agencies may face similar challenges. Specifically, lack of information at relevant scales and budget constraints were cited by both BLM and USFS employees as perceived barriers to adaptation planning (Archie et al. 2012). Furthermore, lack of agency guidance or direction is cited as one of the biggest limitations in prioritizing climate change in land management decisions (GAO 2007). Specific agency direction was a more significant barrier for individuals from the BLM than the USFS (Archie et al. 2012), though we heard from USFS and BLM participants alike that the necessary support and guidance from line officers and decision makers at the planning unit or forest level was currently lacking. Time, funding, and internal and external politics are also barriers to using scientific data and information in land-use planning and management (e.g., Dilling and Lemos 2011, Mukheibir et al. 2013). The managers participating in our study felt their agencies were reluctant to commit time and money to projects when there is uncertainty about the magnitude, timing, or probability of a climate change impact. Finally, several of our study participants felt that climate change was not yet a high profile public issue, and was therefore unlikely to be prioritized within current management. Management priorities are often shaped by public opinion (Archie et al. 2012, Ellenwood et al. 2012), especially because public comment is required by NEPA for any management activity that has potential ecological impacts. Competing priorities may limit how much time resource managers feel they can allocate to training, education, or synthesis of climate change science (GAO 2007, Jantarasami et al. 2010), while also impacting the likelihood that climate change adaptation projects will be funded, implemented, and publically supported.

While recent federal policies guide managers to consider the implications of climate change at all levels of land management planning, most managers we interviewed are not yet thinking about or addressing climate change directly with specific projects. Of the particular management actions addressed in our surveys, participants generally felt that existing management strategies (e.g., thinning, prescribed burning) would be the most effective and likely to be implemented in response to climate change (Figure 4.3). Management actions that are already widely implemented on public lands to meet other objectives are more likely to be supported by decision makers and have relatively little risk of eliciting negative public opinion (GAO 2007), which can be the key to success in a land management agency that must respond to both public input and litigation. For example, former lawsuits that resulted in legal decisions regarding certain management actions may set a precedents that allow managers to know what existing management actions they can take without being formally challenged. Additionally, using existing policies, where applicable, would allow agencies to meet multiple goals without having to necessarily anticipate the future extent and timing of climate-related impacts (Joyce et al. 2009). The Healthy Forest Restoration Act of 2003 (PL 108-148), for example, allows increased forest thinning and prescribed fire to reduce hazardous fuels and wildfire. This policy could be used as support for ongoing and accelerated restoration and fuels treatments that increase forest resilience to disturbance-related impacts of climate change.

Novel adaptation strategies, such as assisted species migration, expanding the genetic stock for revegetation, or managing for future insect and disease outbreaks (e.g., Joyce et al. 2008), on the other hand, were rarely discussed in interviews or workshop discussions, and surveys indicated that most resource managers felt these strategies were unlikely to be implemented by the USFS or BLM. Even though these adaptation strategies might be effective for dealing with climate change, they require anticipation of the timing and extent of future shifts in, for example, species composition or the frequency of extreme events (see Joyce et al. 2009). Many managers recognize the non-linear nature of ecological responses and the stochasticity of disturbance events, which may lead to their reluctance to adopt strategies that rely on future climate and species distribution projections (Joyce et al. 2008). Likewise, extensive monitoring, changes to existing policies and regulations, or adoption of new policies may be required to make novel adaptation strategies a more feasible option (Joyce et al. 2008). For example, although management activities such as assisted migration have been effective in a few trials aimed at eliminating the risk of species extinctions (Joyce et al. 2008) or expanding ranges for commercially valuable timber species (e.g., Willis et al.

2009), there are still tremendous political and ethical ramifications of planting species outside their naturalized range (e.g., Pedlar et al. 2011), and there is little policy guidance when and where this adaptation strategy is appropriate (McLachlan et al. 2007, Schwartz et al. 2012).

Uncertainty about the potential impacts of climate change led many of our participants to focus on general goals or outcomes rather than specific management strategies, such as managing forests and rangelands to be more resilient to future climatic changes. Resilient ecosystems are those that have a greater capacity to gradually respond to climate perturbations or recover more rapidly after disturbance (McLachlan et al. 2007). Although management over the past several decades has often focused on restoring resilience by returning the landscape to historical reference conditions, climate change may necessitate a different approach (Millar et al. 2007, Hobbs et al. 2014). Therefore, guidance is needed to define what ecosystem resilience may look like with potential future changes in climate (West et al. 2009). Basing management decisions on unknown future conditions makes decisions challenging, but proactive adaptive management approaches such as increasing structural and compositional diversity of existing ecosystems, improving connectivity of landscapes for species' migration, and intensive monitoring and treatment after active management are some solutions that have been suggested to allow resource managers flexibility in response to climate change (West et al. 2009). These strategies don't require local-scale or species-specific projections to implement and can be informed by existing ecological knowledge. However, these solutions may only be viable so long as major ecosystem transitions do not occur. Over the long-term, management approaches may need to shift with shifts in ecosystems and resources (Millar et al. 2007, Joyce et al. 2009, West et al. 2009).

Conclusion

Although the science on potential climate change impacts continues to grow and be refined, the types of climate change research resource managers in the USFS and BLM perceive to be available and accessible are not currently effective for creating management prescriptions. However, rather than uniformly increasing the supply of climate science, federal land managers need a process in which they can repeatedly and collaboratively interact with scientists in production and compilation of climate change science that is usable and applicable (Dilling & Lemos 2011). These collaborative efforts could alleviate perceived barriers associated with lack of personnel and resources to develop the information independently (Archie et al. 2014). Federal resource managers desire scale-relevant research focused at sub-regional scales (Archie et al. 2014). Projections that focus on impacts that have direct applicability to management priorities, such as projections about vulnerabilities to fire, flooding, or habitat loss may be perceived as more useful. As peer reviewed journals are not easily accessible or readily used by federal land managers (Archie et al. 2014), having information available on regularly updated websites or blogs could be an important way to ensure its accessibility. Additionally, federal land managers could benefit from workshops, webinars, or trainings that serve as boundary objects for synthesizing relevant information and aim to bridge the research-management gap. The framework for these boundary organizations may already exist in Collaborative Forest Landscape Restoration Programs, Landscape Conservation Cooperatives, and other efforts in place nationally and across the

U.S. northern Rocky Mountain region. These organizations could play an active role in disseminating climate change science, and serve as fertile ground for future research about the effectiveness of boundary objects and organizations.

Having appropriate information is only one part of the challenge of effectively managing for the impacts of climate change. Knowing how to apply that information and having the support and resources to take action are also essential. On public lands managed by the USFS and BLM in the US northern Rocky Mountains, there is a disconnect between mandates at the national level and actions that are being taken at the district or field office level. While national policies for climate change adaptation and mitigation are in place, resource managers still lack the specific guidance and support from decision makers in upper management that would allow them to start managing for climate change impacts. Although there is significant uncertainty associated with managing for climate change impacts, low risk options, such as more widely applying current techniques, may be an easy and effective way to begin to implement climate change adaptation measures on-the-ground (Joyce et al. 2009). These options can be informed by existing regional-scale climate change projections that focus on predictions of potential risks (e.g., to increased frequency of wildfire, flooding). In the short term, focusing on where existing treatments can accomplish multiple goals could reduce costs while stretching limited resources. Adapting existing policies to facilitate climate change adaptation may also allow management flexibility and rapid response measures (Joyce et al. 2009). Collaborative efforts between public, private, and non-profit organizations can increase the suite of viable adaptation options for resource managers by heightening public support and providing guidance on managing more extensive landscapes. Finally, over longer time scales, it will be important to invest in additional research and monitoring on management strategies that are considered potentially effective but are currently not widely implemented as this may increase the probability of their future adoption by agencies.

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Table 4.1. Mean score for each statement about the usefulness of climate change (-3 to +3, strongly disagree to strongly agree) ± 2 standard errors. An ANOVA with Tukey's post hoc tests was performed to determine the statistical significance of differences in mean ratings between each type of information. Superscripts that differ indicate values that differ at alpha = 0.05. N indicates the number of responses to each prompt.

Items	N	Mean ± 2 SE
The global climate change information is useful for land management (modeling and emission scenario information).	60	1.4 ± 0.2^{a}
The regional climate and water research is useful for land management.	61	2.2 ± 0.2^{b}
The regional vegetation and fire research is useful for land management.	59	2.2 ± 0.2^{b}
The local-scale forest vegetation and climate simulations are useful for land management.	58	1.9 ± 0.2^{b}

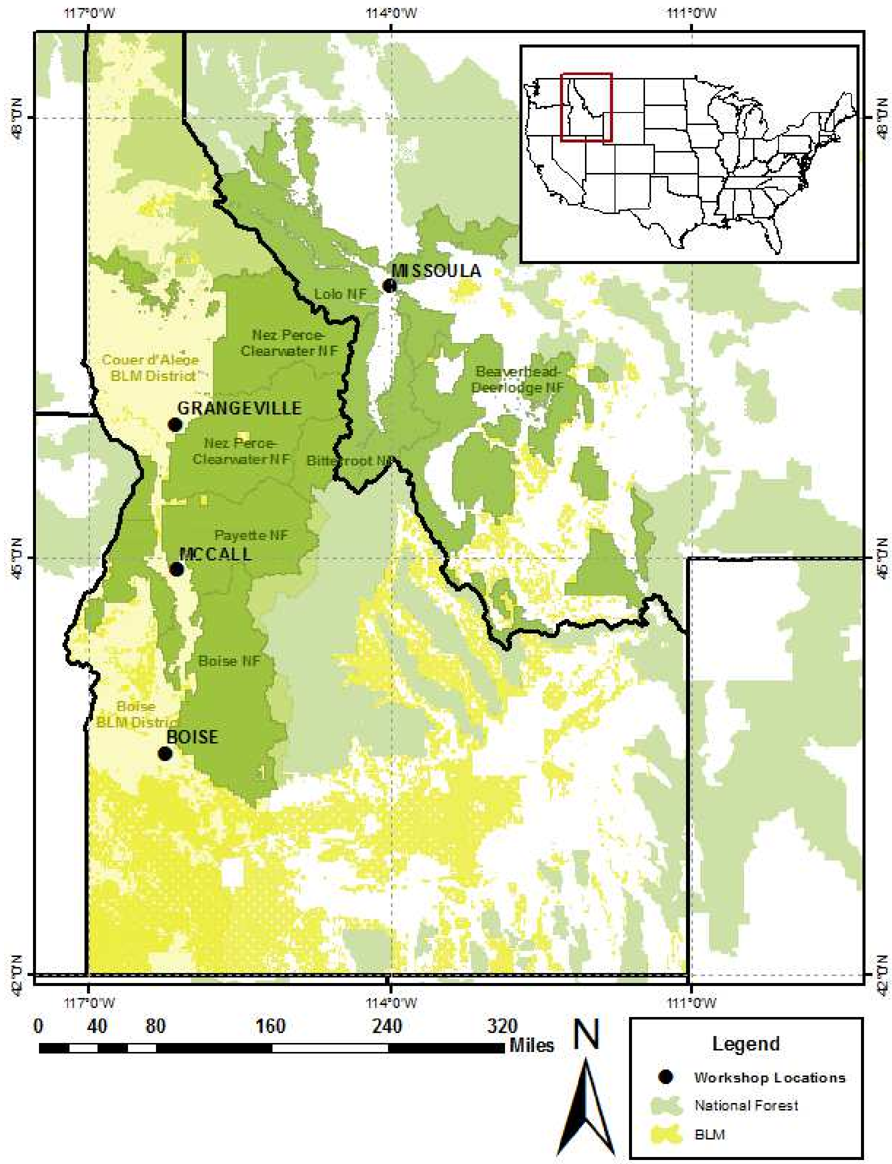

Figure 4.1. Map of the study area highlighting the National Forests (dark green) and Bureau of Land Management (BLM) districts (light yellow) represented by workshop participants. Participants were from six national forests and two BLM districts. The majority of land area in the U.S. northern Rocky Mountains (defined here as Idaho and western Montana) is federal land under the control of the US Forest Service (light green) and BLM (bright yellow).

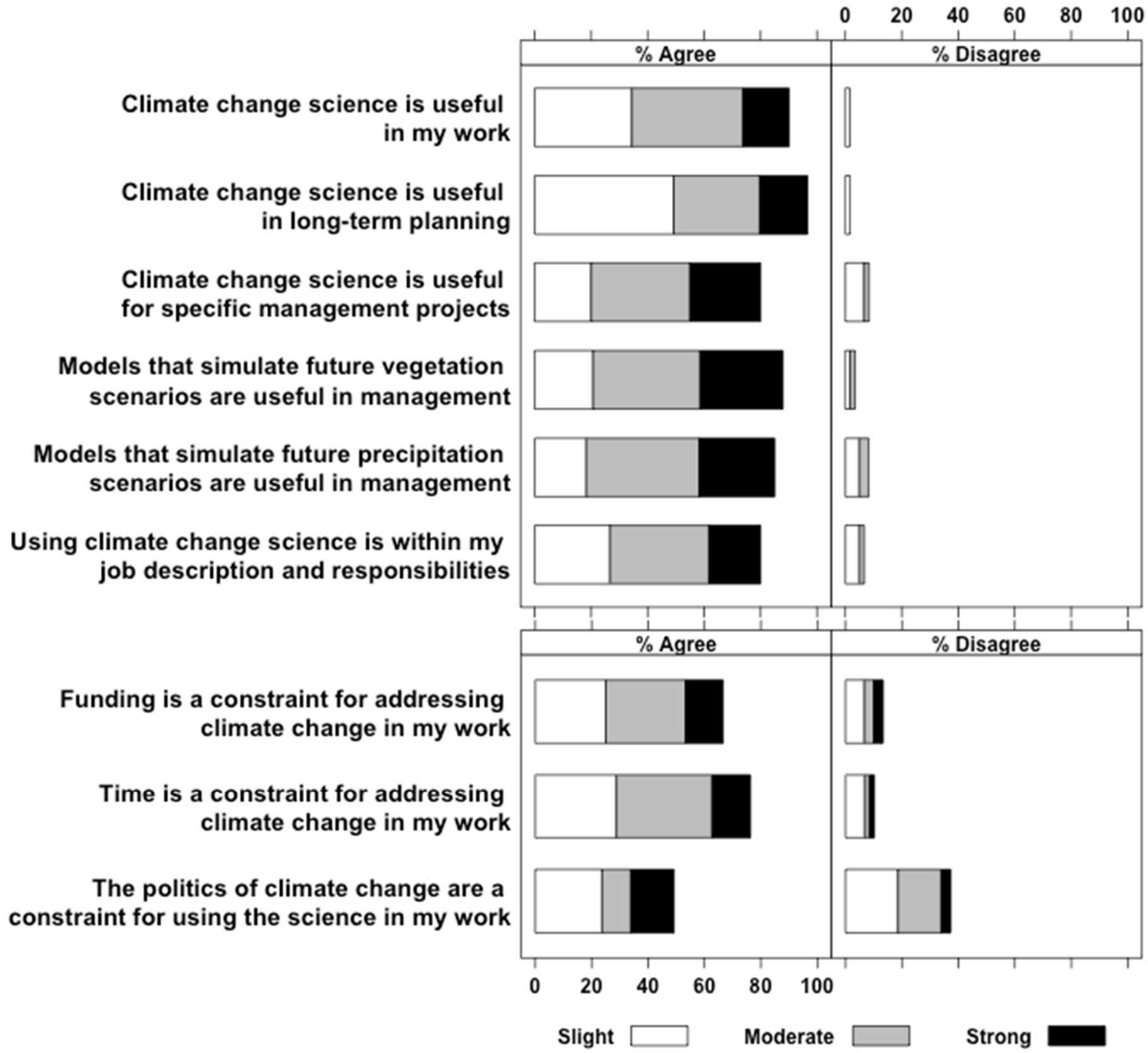

Figure 4.2. Percentage of survey participants that agreed, felt neutral, or disagreed with the statements in the pre- and post- workshop interviews regarding the usefulness of climate change science (top panel) and barriers to using the science to adapt or mitigate the impacts of climate change (bottom panel). The "Agree" column displays the percentage of participants that strongly agree (-3; black bars), agree (-2; grey bars), or slightly agree (-1, white bars) with the listed survey statements. The same is true in the "Disagree" column. Neutral (neither agree nor disagree) responses are not displayed, thus, bars may not sum to 100%.

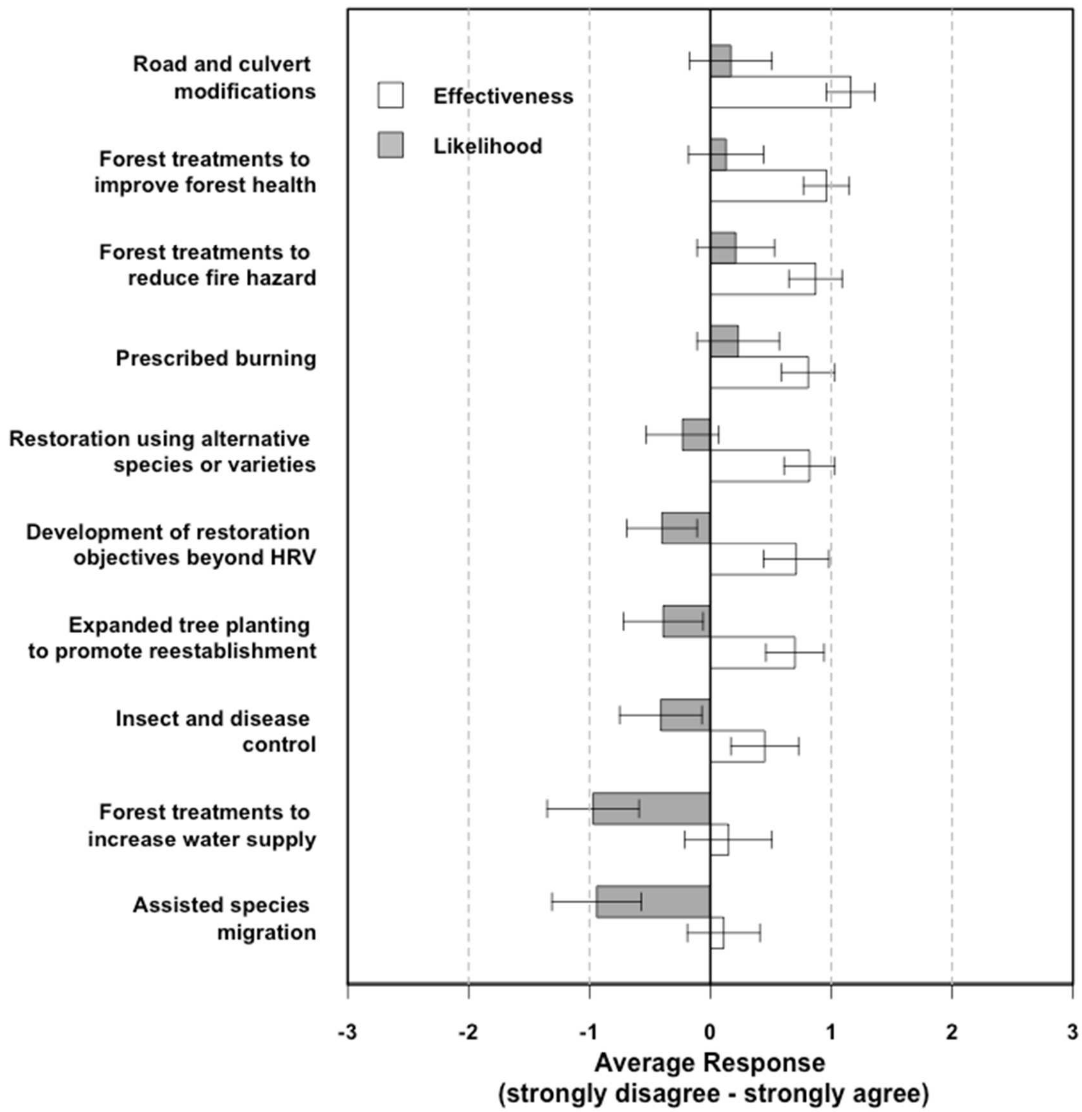

Figure 4.3. Mean response to ten survey questions asking participants (n = 61) to evaluate the efficacy of different management strategies for adapting to climate change in Idaho and Western Montana. Participants were asked to rate whether they felt they actions listed would be effective (white bars) and the likelihood that their agency would use each action (grey bars) to adapt to the impacts of climate change. Responses were scaled from -3 (very ineffective/unlikely) to +3 (very effective/likely). Management actions that were more likely to be considered effective and likely to be implemented in response to climate change are at the top of the figure. Actions that were perceived to be ineffective and have a low likelihood of implementation are at the bottom of the figure. Error bars indicate ± 2 standard errors around the mean. HRV stands for Historical Range of Variability and refers to the range of potential conditions (e.g., disturbance, climate) that a particular ecotype may have experienced prior to European settlement and heavy anthropogenic manipulation of the landscape

CHAPTER 5

Conclusions, Implications, and Research Needs

Considerable concern has been expressed for the future of dry mixed-conifer forests across the west, yet, in dry mixed-conifer forests of the northern Rockies, this research lends support to the high capacity of these forests to retain their structure and function after fire. This results in part from a heterogeneous burn mosaic that creates a variety of patch sizes and burn conditions that leave live seed sources for regeneration. The diversity of species is also important for ensuring the resilience of these forests as a combination of early seral species with different adaptive regeneration traits can colonize a variety of burned patches.

However, as climate conditions get warmer, the burn mosaic will increasingly be more important for driving patterns of mortality than regeneration. Even where seed sources are available, high summer temperatures will constrain germination and survival of post-fire regeneration. As disturbances continue to become more frequent and extensive, it will be increasingly likely that important thresholds of resistance to change are passed (Millar and Stephenson 2015). The balance between tree mortality and tree regeneration will determine if and how future fires mediate contractions in dominant dry mixed-conifer forest species. If proportionally more area burns in stand-replacing fires, for example, and much of the dry mixed-conifer forest ecotype in the northern Rockies does not regenerate, upslope shifts in the cover type will occur and forests may be replaced by grassland or shrubland communities. Results from Chapter 3 highlight, for example, that within 50 years, more than 60% the dry mixed-conifer forest zone may experience significant decreases in seedling regeneration where fires have removed mature trees. Where trees survive fire or do not experience disturbance, upslope shifts in species composition may occur over a much longer time frame.

Dry mixed-conifer forests have been extensively managed in the past and will continue to be managed for the provision of important ecosystem services, economically viable products, and critical habitat in the future. Past management has impacted the resilience of these forests while the actions that managers propose to deal with the impacts of climate change will inevitably affect the future resilience of these forests. Forest managers are most likely to use existing management strategies to adapt to the potential impacts of climate change, but data that support species-specific responses to potential climate warming may help managers explore more novel techniques, such as prioritizing where and what they plant to be adapted to future climate conditions.

Implications for management

Regeneration and rehabilitation after wildfire are a high priority for resource managers. Results from this dissertation help highlight areas of the landscape where management intervention intended to insure successful reforestation could be effective, allowing forest managers to strategically focus and prioritize their activities. For example, as summer temperatures continue to rise, increasingly large areas of the landscape will become unfavorable for seedling regeneration, resulting in low survival of planted seedlings, while other areas of the landscape will have successful natural regeneration and not require any intervention. Managers might therefore be able to focus their reforestation efforts on areas where the climate or microenvironment remains favorable but a seed source limitation needs to be overcome. This research not only helps resource managers prioritize the sites that have potential for regeneration of species of interest but also suggests areas where managers might allow natural transitions to occur because these sites are likely to become unfavorable for tree establishment in the recent future.

Locally or regionally specific information that identifies important thresholds and predicts future changes on landscapes, such as the research detailed in Chapters 2 and 3, will help forest managers anticipate change by giving them ecologically identifiable targets, effectively create management prescriptions, and plan projects that better account for climate change. The results of Chapter 4 clearly articulate the explicit need of forest managers for regional and local climate-change projections. In Chapter 3, I use regionally collected observational data to build and inform statistical predictions of species distribution shifts in response to climate and fire in the northern Rockies. Studies that link empirical data with statistical models at this scale will likely help managers better understand field observations and develop actions for dealing with observed and future impacts of disturbance and climate change. Chapter 3 also incorporates local variations in environmental gradients, topography, and disturbance in our predicted response, better accounting for variability that otherwise could not be effectively modeled or predicted at larger spatial scales. This is important, as managers repeatedly emphasized during interviews that they felt the uncertainties inherent in fine scale process-based models made them reluctant to use them as a basis for proposed management actions. Understanding climate and landscape thresholds for forest regeneration also allows managers use current management techniques until significant ecosystem changes necessitate adaptation (Groffman et al. 2006, Millar et al. 2007).

Future research needs

An assumption that pervades much of the thinking about restoration of dry mixed-conifer forests is that these forests are so far departed from historic conditions that current fires are burning in an uncharacteristically severe manner (Covington 2000). Identifying whether this assumption is valid is important for assessing resilience to fires in these forests. For example, research in Chapter 2 emphasizes the importance of nearby live seed sources for tree regeneration. Large stand-replacing patches, if they are present or increasing, could therefore have important implications for forest recovery. Several researchers have attempted to quantify whether area burned severely has changed in recent decades (e.g., Miller et al. 2009, Dillon et al. 2011, Cansler and McKenzie 2013, Baker 2015), but there is currently no consistent pattern emerging across the western U.S. Some regions exhibit increases in the proportion of area burned severely and other regions lack detectable change. Longer burn severity records, where available and interpretable from aerial photography (e.g., Hessburg et al. 2007) or other methods (see Odion et al. 2014, Baker 2015), may help us interpret changes in fire severity through time, especially in systems where fire intervals are variable and may be up to or exceed 30 years (the current temporal extent of satellite data). Likewise, as satellite-derived burn severity data continue to become increasingly available, we may be better able to quantify changes in severity and stand-replacing patch sizes to predict whether burn severity will decrease the resilience of these dry forest ecosystems. Additionally, there is a need to understand how burn severity patterns and patch dynamics differ in forests with a history of land management and fire suppression versus areas where the natural role of fire has been maintained. Whether or not these patterns differ as a result of management history could provide important context for forest restoration efforts by helping elucidate whether current fires are departed from conditions that might otherwise occur or if current fires are restorative of natural properties of the ecosystem (regardless of management history).

Furthermore, few studies have examined the relationships between interannual climate and tree seedling regeneration. Tree species at the lower margin of their distribution are likely to be more susceptible to range contractions with future climate change (e.g., Bell et al. 2014), and, thus, it will be important to understand the physiological and climatic mechanisms responsible for seedling establishment and failure along ecotone boundaries. Although gradual changes in climate will eventually compound to impact post-disturbance forest recovery, year-to-year variability in climate and the episodic nature of extreme events will likely be more important for determining whether seedlings can get established in the immediate future (League and Veblen 2006, Jackson et al. 2009). Annual climate drivers of success or failure of seedling establishment could therefore be important for understanding the implications of stochastic events on forest resilience to future fire.

Finally, long-term repeated measures studies that track successional dynamics and change after disturbance events will be essential for understanding how disturbance and climate change interact to influence the distribution and composition of forest species. Few studies successfully track changes through time, but where historical data exist and can be re-measured, these data may give us important information about shifts in species distributions over time (e.g., Brusca et al. 2013) or provide important information about patterns and processes that interact to influence ecological responses to disturbance (Halofsky et al. 2011, Romme et al. 2011). Additionally, our understanding of ecological turnover between life stages of key tree species (sensu Lloyd et al. 2005) in the northern Rockies is lacking, limiting our ability to predict the future consequences of regeneration patterns that exist currently on the landscape. For example, we can only surmise how much regeneration is necessary to transition a site from seedlings to mature trees, and it is likely that these transition probabilities will continue to change and be impacted by warmer and drier climate conditions. Studies that are established immediately following fire and resampled annually would help elucidate these dynamics.

Understanding the diverse and interacting ecological factors that influence the recovery or decline of dry mixed-conifer forests will be increasingly important to accurately predict the impacts of future disturbance and climate change. Although this dissertation begins to examine potential mechanisms and drivers of seedling regeneration dynamics, continuing to advance our understanding of the nature and likelihood of changes in burn severity and stochastic extreme events, while recording empirical observations through time, will help improve our predictions about the interacting and mediating roles of climate and fire on one another. How forest managers decide to deal with potential changes in dry mixed-conifer forests will be tempered by social values and perceptions. Understanding how these values and perceptions shape management priorities will continue to be important for effectively managing ecosystems through transitions and climate change.

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Appendix B: Supplementary Methods for Chapter 2

Defining the range of climate in dry-mixed conifer forests

We obtained derived actual evapotranspiration (AET) and potential evapotranspiration (PET) climate data at 4 km resolution for the entire study region from 1979 to 2010 (Abatzoglou 2013). AET is the actual amount of water lost from the environment from evaporation from the soil surface and transpiration from the plant canopy. PET is the maximum potential amount of evapotranspiration that can occur given that an ecosystem is not water limited. The ratio of these two variables (AET/PET) represents the amount of moisture available for plants to use during the growing season, and therefore may be a more direct measure of the impacts of climate (e.g., temperature and precipitation) on plant establishment, survival, and growth. The ratio of AET/PET can range from 0 to 1 where low values typically correspond to warmer and drier environments and values close to 1 typically correspond to cooler and wetter environments. Furthermore, many biomes can be differentiated along axes of evaporatranspiration and water deficit (e.g., Stephenson 1990), making the AET/PET a good measure of differences in vegetation types across broad geographic regions. We derived climatologies for the entire study region by averaging monthly values of AET and PET over the 32-year period from 1979 to 2010. Each pixel defined as dry-mixed conifer forest was assigned a climate value from the climatology surface and used to characterize the distribution of AET/PET over the entire cover type. We also assigned an AET/PET value for each of our sites. We compared the distribution of these values with the distribution of values for dry mixed-conifer forests to ensure that our sampling covered the full distribution of possible climatic characteristics where these forests are found.

Goodness-of-fit tests for logistic regression models

To evaluate the goodness of fit of each of our logistic regression models, we used three summary measures: the deviance residual, a Hosmer-Lemeshow test, and the area under the curve (AUC) of a receiver operating characteristic (ROC) curve. The model deviance is estimated as the maximum likelihood ratio of a saturated model versus a fitted model, the test statistic of which is distributed as a Chi-squared distribution (Hosmer et al. 2013). However, as the number of parameters in the saturated model approaches the number of samples, n, the estimated p-values from the chi-squared distribution can be incorrect (Hosmer et al. 2013). One method to avoid this issue is to grouped the probabilities of the fitted model into a number of equal groups (we used ten) based on percentiles of the data and compared the observed and expected frequencies of observations in each group using a Hosmer-Lemeshow test (Hosmer and Lemeshow 1980). The Hosmer-Lemeshow test uses a Pearson's goodness-of-fit statistic:

Eqn. 1

$$\sum_{i=0}^{1} \sum_{j=1}^{g} \frac{(o_{ij}-e_{ij})^2}{e_{ij}}$$

where o_{0j} denotes the number of observed $Y = 0$ observations in the j^{th} group, o_{1j} denotes the number of observed $Y = 1$ observations in the j^{th} group, and e_{0j} and e_{1j} similarly denote the expected number of zeros in each group respectively (Hosmer and Lemeshow 1980, Hosmer et al. 2013). The Pearson's test statistic approximates a Chi-squared distribution with the degrees of freedom equal to the number of groups (g) minus two (Hosmer and Lemeshow 1980). The model fit statistics indicate a significant lack of fit to the data if the p-value generated from the deviance chi-squared test or Hosmer-Lemeshow statistic is < 0.05. Thus, failure to reject the null hypothesis of a significant lack of fit provides evidence in support of good model fit to the data.

For binary models indicating the presence or absence of an event, an ROC curve can also be used to evaluate model fit. The ROC curve plots the tradeoffs between correctly classified (true positives) and incorrectly classified (false positives) presences for every possible classifier threshold. The point at which the true positives are maximized relative to the false positives can be identified as an optimal classifier probability (Fawcett 2006). Additionally, to compare the performance of classifier values among models, the area under the ROC curve (AUC) can be calculated. The AUC is equal to the probability that a randomly chosen positive instance will be ranked higher than a randomly chosen negative instance (Fawcett 2006). Thus, higher AUC values indicate better the model performance. AUC values can range from 0.5 (i.e., no better than random) to 1 (i.e., perfect ability to discriminate between positive and negative instances).

Finally, to determine if models were overfit, we developed cross-validation techniques to evaluate the Hosmer-Lemeshow statistic and AUC for each logistic regression model. A random sample (without replacement) of 80% of the data were used to train the models, and the remaining 20% of the data were used to validate the models (as suggested by Hosmer et al. 2013). Each model was bootstrapped 1000 times using a different subset of the data. Medians and 95% confidence intervals were summarized for each of the validation statistics.

Generalized linear models for count data

Count data are inherently discrete and positive-skewed (non-negative), thus, several unique distributions are commonly used to model counts, including the Poisson, quasi-

Poisson, and negative binomial distributions. The most common distribution for modeling count data is a Poisson, where the mean, μ_i, is equal to the variance. However, in most ecological datasets the variance is larger than the mean, causing overdispersion in the data and limiting the utility of the Poisson distribution. If model dispersion >> 1, the dataset must be corrected for overdispersion by refitting the model with either a quasi-Poisson model, which adjusts the variance by including a dispersion parameter $Var(Y_i) = \varphi \, \mu_i$, or a negative binomial model, which is a mixture of the Poisson and gamma distributions. Overdispersion can be caused by model misspecifications, such as missing covariates, interactions, or outliers in the response variable, clustering of observations, excess zeros, or variation in the data that is actually much larger than the mean (Hilbe 2011).

If some or all of the overdispersion in a model is due to excess zeros, and the number of zeros exceeds that predicted by the Poisson or negative binomial models, hurdle or zero-inflation models can be considered. Hurdle models separate the process of generating zeros and non-zeros, where a binomial model is used to predict the probability of a zero observation, and a truncated Poisson or negative binomial distribution models the non-zero observations (Martin et al. 2005, Zuur et al. 2009, Hilbe 2011). These models assume a non-zero count can only be measured if some ecological "hurdle" has been surpassed, and once it has been passed, there is no chance of a zero occurrence (Martin et al. 2005, Zuur et al. 2009). Zero-inflated (ZI) models, in contrast, specify the zeros as resulting from two different processes, the binomial process and the count process. The binomial portion of the ZI model predicts the presence of false zeros (e.g., zeros that occur as a result of design error, observation error, or sampling outside of the habitat range of that species) and the count portion of the model allows for both zeros and positive counts (Martin et al. 2005, Zuur et al. 2009).

We considered both hurdle and zero-inflated models, but ultimately choose not to model our data with hurdle models because even if a so-called ecological hurdle was surpassed and a seedling established on a site, it could still die after establishment, resulting in an absence of its count. Instead, we used zero-inflated models to represent the abundance of a species on a site. *False* zeros in our data could have resulted from several alternative scenarios: our sampling area was not large enough to represent the patch as a whole (e.g., randomly placed but potentially biased), we sampled outside a species' optimal habitat (e.g., for rarer species like lodgepole pine, which may be more typical in subalpine forests), or tree seedlings that were present were missed in the sampling effort. We attempted to account for sample size bias by increasing our sampling effort in locations where few seedlings were noticed prior to starting our sampling effort. However, because we could not represent the other two sources of potential false zeros in the zero-portion of the model, and because we assumed that the probability of each of these errors was relatively constant across sites, we decided that an intercept-only predictor function would be sufficient for the zero-portion of the model (Zuur et al. 2012). If the probability of false zeros in a zero-inflated negative binomial model is low, the model converges to a negative binomial.

To find the optimal count model we compared the fit of the Poisson (P), quasi-Poisson (QP), negative binomial (NB), zero-inflated Poisson (ZIP), and zero-inflated negative binomial (ZINB) for count of tree seedlings of all species combined and the four most abundant species on our sites. Models were compared using AICc, which is corrected for the number of parameters, k, in a particular model and Pearson's dispersion test, where

Pearson's dispersion statistic, φ, is defined as:

Eqn. 2 $$\varphi = \Sigma \frac{(R_i^P)^2}{(n-k)}$$

n indicates the sample size (e.g., number of sites), k is the number of estimated parameters included in the model, and R_i^P are the Pearson's residuals, estimated here as:

Eqn. 3 $$R_i^P = \frac{y_i - \hat{y}_i}{\sqrt{V(\hat{y}_i)}}$$

where $\hat{y}_i$ is the predicted value and $V(\hat{y}_i)$ is the variance of the ith observation of y_i (Hilbe 2011). Values of $\varphi > 1$ indicate that model may be overdispersed or misspecified, whiles values $\varphi < 1$ indicate underdispersion. Candidate models were compared using a cutoff of ΔAICc ≤ 2 and a $\varphi \approx 1$ (Table A1). Models were further selected using a Vuong likelihood ratio test to determine the best-fit model of each of the candidate models (Vuong 1989). Additionally, we compared the predicted number of zeros and mean counts from each distribution with the observed number of zeros and mean counts (Zeileis et al. 2008).

Each model displayed overdispersion, and Vuong likelihood ratio tests confirmed that a negative binomial or zero-inflated negative binomial distribution was better than their Poisson counterparts for each model. For the all species and Douglas-fir models, a negative binomial model was the best fit of the five possible candidate models. A zero-inflated negative binomial model was the most parsimonious model for predicting the abundance of ponderosa pine, lodgepole pine, and grand fir. Because these three species were present on significantly fewer sites than Douglas-fir (65% of transects for Douglas-fir; 32, 26, and 15% of all transects, respectively, for the other species), there was an excess of zeros in the count, thus a zero-inflated model helped correct for some of the overdispersion present in the Poisson model, as well as convergence problems due to underdispersion in the negative binomial models.

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Table B1. Fit statistics for each of the exponential family count models. The optimal model is indicated by an asterisk. Models were selected based on AICc criteria and the closeness of the dispersion parameter to 1.

	# of zeros	Mean	AIC	AICc	Dispersion Parameter
		All Species			
Observed	48	54.7	-	-	-
Poisson	7	54.7	7526.87	7528.47	41.37
Quasi-poisson	NA	54.7	NA	NA	NA
Negative Binomial*	38	61.3	1412.52	1414.12	1.20
ZIP	48	54.6	6361.81	6369.70	5.56
ZINB	38	61.3	1414.53	1417.12	1.23
		Douglas-fir			
Observed	63	22.7	-	-	-
Poisson	23	22.7	5090.20	5091.80	27.40
Quasi-poisson	NA	22.7	NA	NA	NA
Negative Binomial*	58	29.8	1111.62	1113.22	1.11
ZIP	63	22.9	4331.55	4339.44	21.36
ZINB	58	22.8	1113.62	1116.21	1.17
		ponderosa pine			
Observed	123	3.7	-	-	-
Poisson	55	3.7	1770.96	1772.56	9.42
Quasi-poisson	NA	3.7	NA	NA	NA
Negative Binomial	122	5.7	557.08	558.68	0.67
ZIP	123	3.7	1052.31	1060.20	3.13
ZINB*	122	4.6	554.71	557.30	0.95
		lodgepole pine			
Observed	129	14.9	-	-	-
Poisson	53	14.9	3133.58	3135.18	17.50
Quasi-poisson	NA	14.9	NA	NA	NA
Negative Binomial	124	41.5	608.78	610.38	0.61
ZIP	129	18.4	1694.59	1702.48	6.12
ZINB*	124	53.2	609.17	611.76	1.23
		grand fir			
Observed	151	6.5	-	-	-
Poisson	79	6.5	1891.60	1893.20	10.57
Quasi-poisson	NA	6.5	NA	NA	NA
Negative Binomial	149	12.3	379.21	380.81	0.35
ZIP	151	6.7	644.81	652.70	2.05
ZINB*	149	11.3	377.15	379.74	0.86

Appendix C: Supplementary Results for Chapter 2

Cross validation of logistic and negative binomial models

Cross validation indicated that the logistic regression models for seedling presence and absence were more robust than the negative binomial models of seedling counts (Table C1). For the all species logistic regression model, a median AUC value from cross-validation of 0.762 suggested that this model was not overfit. Furthermore, all of the species-specific logistic regression models cross-validated well except for the grand fir model (Table C1). Because the grand fir model is potentially overfit, its results may not be widely applicable outside of our study region. For the negative binomial count models, the medians of 1000 cross-validation runs for the Pearson's chi-squared statistic indicated a significant lack of fit between the data and the model distribution for three of the five models (Table C1). This implies that these models were prone to overfitting and therefore, are likely to have poor predictive performance. We therefore suggest caution in generalizing the results of these models to dry mixed-conifer forests outside of our study region.

Table C1. Cross-validation statistics for each of the logistic and negative binomial models. Values for χ^2 and p are medians of the 1000 model runs with 95% confidence intervals reported in brackets for the AUC and Pearson's Chi-squared validation statistics. AUC values > 0.5 indicate the model's ability to discriminate between presences and absences is better than random. The Hosmer-Lemeshow and Pearson's chi-squared statistics indicate a significant fit of the data to the model distribution if p-values > 0.05. The Pearson's statistic for the negative binomial model is based on Monte Carlo resampling (n =1000) of the data for each run.

	Logistic Regression			Negative Binomial GLM	
		Hosmer-Lemeshow		Pearson's statistic	
Model	AUC [95% CI]	χ^2	p	χ^2 [95% CI]	p
All Species	0.762 [0.571 - 0.934]	13.865	0.085	160.1 [154.4 - 163.2]	0.025
Douglas-fir	0.795 [0.650 - 0.904]	15.439	0.051	387.8 [122.9 - 1554.0]	0.043
ponderosa pine	0.790 [0.656 - 0.899]	12.769	0.120	97.3 [19.3 - 320.0]	0.060
lodgepole pine	0.795 [0.639 - 0.917]	12.525	0.129	686.5 [76.2 - 4761.2]	0.020
grand fir	0.750 [0.546 - 0.941]	17.391	0.026	227.1 [76.2 - 4761.2]	0.070

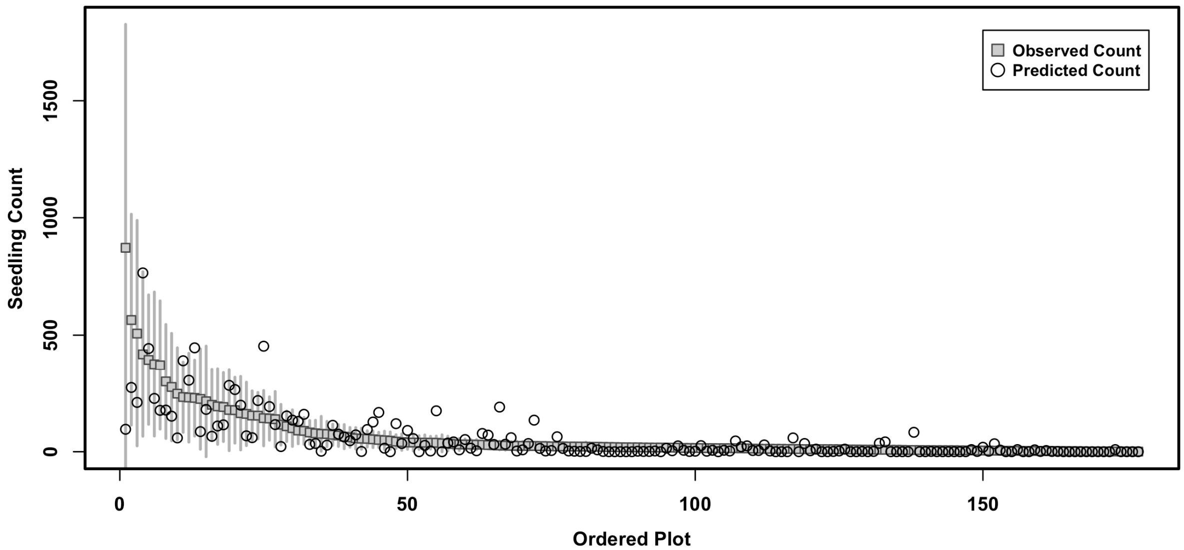

Figure C1. Negative binomial model results. Observed (filled squares) and predicted (open circles) counts with two standard errors (bars) around the predicted count estimates from the negative binomial all-species model. Observations are ordered on the x-axis from lowest to highest predicted count.

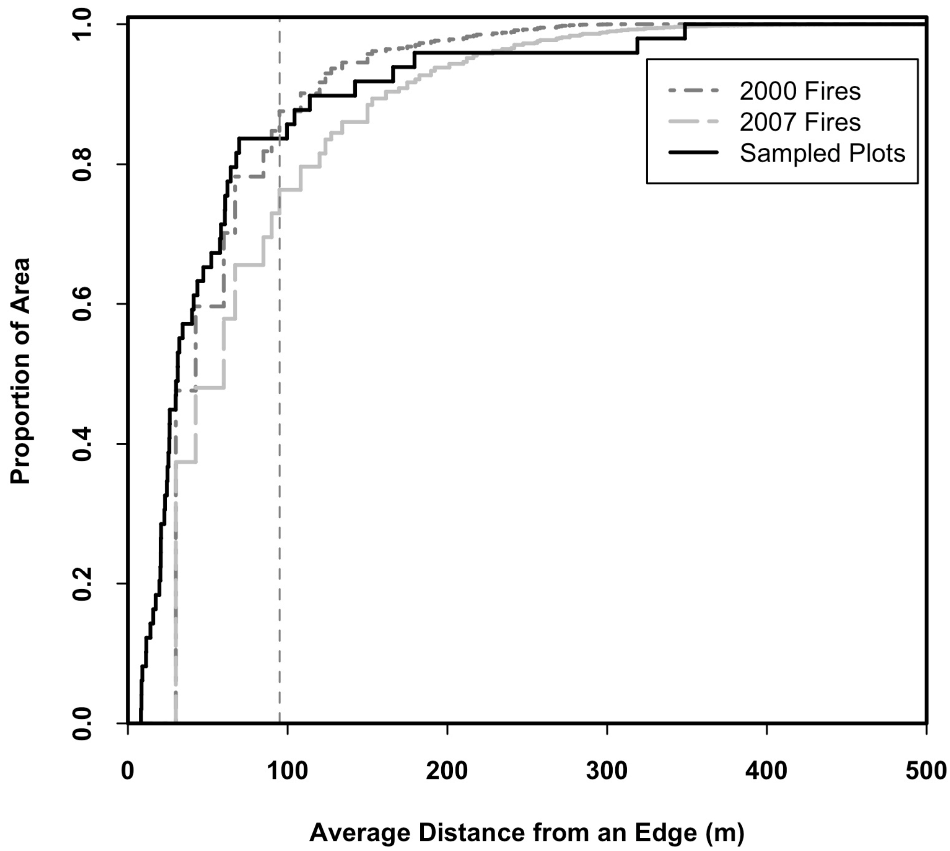

Figure C2. Cumulative distribution functions (CDF) for the proportion of stand-replacing patches in fires from 2000 or 2007. The CDF for the sampled sites is derived from the average distance to a live seed source for all sites sampled in stand-replacing patches, as measured on the ground and with a maximum value of 500 m. The vertical dashed line corresponds to the threshold distance of 95 m, identified using the logistic regression model for all species. Eighty-five percent of the sites sampled were within this threshold distance, while 85% and 75% of the area burned in high severity patches in 2000 and 2007, respectively, were also within 95 m of a lower severity edge.

Appendix D: Selection Criteria for GAM Candidate Models

Table D1. Selection criteria for candidate models were based on AIC and the percent of explained deviance for each model. Models with the lowest AIC values and highest percentage explained deviance were compared by examining model residuals versus fitted values. The models with the best residual patterns were selected for further model reduction and fitting. Best models for each species are indicated by a star in front of the AIC value.

	Douglas-fir		**ponderosa pine**	
Generalized additive model formula	AIC	% Exp. Deviance	AIC	% Exp. Deviance
abun ~ log(area) + s(aet) + s(seasonality) + s(dist) + s(rdnbr) + s(canopycov)	1133.75	40.1	561.93	27.8
abun ~ log(area) + s(aet) + s(seasonality) + s(dist) + s(rdnbr) + s(ltsd)	1129.12	41.6	561.76	28.1
abun ~ log(area) + s(aet) + s(summertemp) + s(dist) + s(rdnbr) + s(canopycov)	1128.61	42.4	558.01	31.7
abun ~ log(area) + s(aet) + s(summertemp) + s(dist) + s(rdnbr) + s(ltsd)	* 1127.56	43.8	557.48	32.7
abun ~ log(area) + s(aetpet) + s(dist) + s(rdnbr) + s(canopycov)	1136.78	37.6	565.45	22.4
abun ~ log(area) + s(aetpet) + s(dist) + s(rdnbr) + s(ltsd)	1134.36	39.2	565.23	22.8
abun ~ log(area) + s(h2odef) + s(summertemp) + s(dist) + s(rdnbr) + s(canopycov)	1129.57	42.0	555.68	33.2
abun ~ log(area) + s(h2odef) + s(summertemp) + s(dist) + s(rdnbr) + s(ltsd)	1128.13	43.7	* 554.28	33.5
abun ~ log(area) + s(springppt) + s(soilmoisture) + s(summertemp) + s(dist) + s(rdnbr) + s(canopycov)	1130.98	44.0	557.12	32.8
abun ~ log(area) + s(springppt) + s(soilmoisture) + s(summertemp) + s(dist) + s(rdnbr) + s(ltsd)	1129.73	44.2	555.66	33.8
abun ~ log(area) + s(summerppt) + s(soilmoisture) + s(seasonality) + s(dist) + s(rdnbr) + s(canopycov)	1132.63	41.5	561.46	30.2
abun ~ log(area) + s(summerppt) + s(soilmoisture) + s(seasonality) + s(dist) + s(rdnbr) + s(ltsd)	1128.00	42.7	562.06	33.7
abun ~ log(area) + s(summerppt) + s(soilmoisture) + s(summertemp) + s(dist) + s(rdnbr) + s(canopycov)	1130.64	42.6	557.83	32.5
abun ~ log(area) + s(summerppt) + s(soilmoisture) + s(summertemp) + s(dist) + s(rdnbr) + s(ltsd)	1128.58	44.3	555.78	33.8

Explanatory variable names are as follows: Annual AET, aet; AET/PET, aetpet; Annual water deficit, h2odef; June through September soil moisture, soilmoisture; June through September temperature, summertemp; Seasonality, seasonality; March through May precipitation, springppt; June through September precipitation, summerppt; Distance to seed source, dist; Burn severity, rdnbr; Canopy cover, canopycov; Live tree stand density, ltsd

Appendix E: Supplementary Results for Chapter 3

Table E1. Parameter estimates and expected degrees of freedom for the original full Douglas-fir model and the reduced model. Model terms were eliminated using a backwards selection procedure until dropping terms no longer resulted in a significant improvement in model fit (Δ AIC $\geq$ 2). Random effects for the time since fire (i.e., the number of years that transpired between the fire and sampling) and fire event as well as interaction terms were added to improve model residuals.

Douglas-fir full model (AIC = 1127.56)				
Parametric coefficients	**Estimate**	**Std. Error**	**z value**	**p-value**
Intercept	8.60	1.00	8.58	< 0.001
log(area)	-1.10	0.17	-6.56	< 0.001
Smoothed terms	**EDF**	**Reference DF**	χ^2	**p-value**
s(aet)	1.00	1.00	1.16	0.28
s(summertemp)	3.32	4.16	13.02	0.01
s(distPSME)	1.00	1.00	19.67	< 0.001
s(rdnbr)	2.29	2.89	4.57	0.19
s(ltsd)	2.24	2.79	5.57	0.12

Douglas-fir reduced model (AIC = 1088.10)				
Parametric coefficients	**Estimate**	**Std. Error**	**z value**	**p-value**
Intercept	12.35	1.10	11.27	< 0.001
log(area)	-1.11	0.16	-7.00	< 0.001
ldist	-1.06	0.15	-7.19	< 0.001
ltsd	0.00	0.00	-2.72	0.01
Smoothed terms	**EDF**	**Reference DF**	χ^2	**p-value**
te(summertemp)	3.00	3.50	17.12	< 0.001
te(fire.id)	10.97	20.00	36.16	< 0.001
ti(summertemp,ldist)	1.00	1.00	21.64	< 0.001

Table E2. Parameter estimates and expected degrees of freedom for the original full ponderosa pine model and the reduced model. Model terms were eliminated using a backwards selection procedure until dropping terms no longer resulted in a significant improvement in model fit (Δ AIC $\geq$ 2). Random effects for the time since fire (i.e., the number of years that transpired between the fire and sampling) and fire event as well as interaction terms were added to improve model residuals.

Ponderosa pine full model (AIC = 554.81)				
Parametric coefficients	**Estimate**	**Std. Error**	**z value**	**p-value**
Intercept	-2.11	2.01	-1.05	0.30
log(area)	0.29	0.29	1.01	0.31
Smoothed terms	**EDF**	**Reference DF**	**χ^2**	**p-value**
s(aetpet)	1.00	1.00	5.83	0.02
s(summertemp)	3.74	4.67	24.33	< 0.001
s(distPIPO)	2.69	3.42	1.61	0.73
s(rdnbr)	1.00	1.00	2.59	0.11
s(canopycov)	1.00	1.00	0.55	0.46

Ponderosa pine reduced model (AIC = 529.89)				
Parametric coefficients	**Estimate**	**Std. Error**	**z value**	**p-value**
Intercept	-2.70	1.71	-1.58	0.12
log(area)	0.36	0.28	1.31	0.19
Smoothed terms	**EDF**	**Reference DF**	**χ^2**	**p-value**
te(summertemp)	2.55	3.10	9.92	0.02
te(tsf)	0.00	1.00	0.00	0.47
te(fire.id)	13.52	20.00	54.89	< 0.001

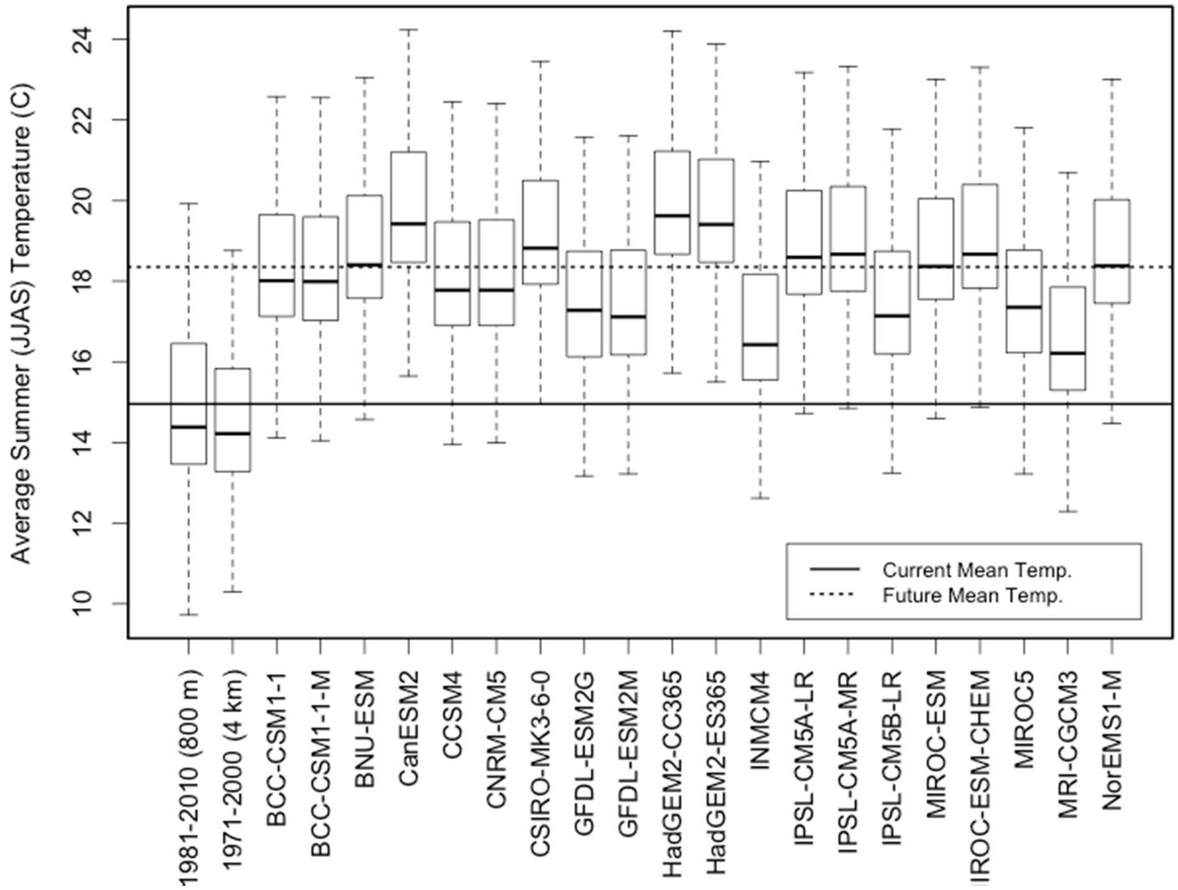

Figure E1. Mean summer temperatures for all sample sites for the historic period (1971-2000 and 1981-2010) and the future (2041-2070) for 20 different global circulation models. Boxplots represent the median, 25th, and 75th percentiles and the whiskers extend to the 10th and 90th percentiles of the data. Horizontal lines display the current (solid) and future (dashed) mean temperatures.

Appendix F: Copyright from *Ecology and Society* Journal

The journal *Ecology and Society* is open access and therefore, I retain all rights to the publication and reprint of this work.

Appendix G: Institutional Review Board Project Approval

University of Idaho

Office of Research Assurances
Institutional Review Board
PO Box 443010
Moscow ID 83844-3010

Phone: 208-885-6162
Fax: 208-885-5752
irb@uidaho.edu

October 3, 2012

To: Hall, Troy
Cc: Blades, Jarod

From: Traci Craig, PhD
Chair, University of Idaho Institutional Review Board
University Research Office
Moscow, ID 83844-3010

Title: 'Multi-scale Climate Change Information for Forests of the Northern Rockies'

Project: 12-303
Approved: 10/02/12
Expires: 10/01/13

On behalf of the Institutional Review Board at the University of Idaho, I am pleased to inform you that the protocol for the above-named research project is approved as offering no significant risk to human subjects.

This approval is valid for one year from the date of this memo. Should there be significant changes in the protocol for this project, it will be necessary for you to resubmit the protocol for review by the Committee.

Traci Craig

Traci Craig

Appendix H: Pre- and Post-Workshop Interview Questions

Interview questions asked to participants before and after the workshops. All questions were open-ended, allowing for a range of responses from participants.

Pre- workshop interview questions

- Do you use climate change science in the work you do? How?
- Other than personal use, is your organization currently using science about climate change impacts? How?
- Tell me what you think about the usefulness of climate change science in the work you do. What makes it useful or impedes its usefulness?
- Are there organizational barriers that impede usefulness?
- Are you aware of forest management actions that could reduce climate change impacts? (e.g., Specific on-the-ground actions)
- Are any of these actions being done now? Why or why not?
- How confident do you feel in the ability of your organization/agency to take actions to reduce the potential impacts of climate change? Will they do it?
- Do have anything else you would like to add about what we have discussed today?

Post-workshop interview questions

- How useful is the climate change science and tools we presented at the workshop for the work you do? What makes it useful or impedes its usefulness?
- Based on the information presented at the workshops, how confident do you feel in the ability of your organization/agency to take actions to reduce the potential impacts of climate change? Will they do it?

Appendix I: Pre-Workshop Survey

Opinions about Climate Change Science, Impacts, and Forest Management

Conducted by the Northern Rockies
Interdisciplinary Research Team

University of Idaho
College of Natural Resources

Please be assured that all answers provided are confidential. This research has been reviewed and approved by the University of Idaho Institutional Review Board.

To maintain your confidentiality, but allow us to match your pre-workshop survey and post-workshop survey with a unique ID, please enter the **last two letters of your first name, the year of your birth, and the workshop location you are attending**.

Last 2 letters of your first name:__________

Year of your birth (YYYY):_________

Workshop location you are attending:___________

Section A: Credibility and Usefulness of Climate Change Science

We are interested in your opinions about the credibility (accuracy/validity) and usefulness of climate change science in land use planning and management projects. For each of the following questions, please select the answer that most closely reflects your opinions.

Question 1. Please indicate your level of agreement with the following statements:

	Strongly Disagree			Neutral			Strongly Agree	Don't Know
Climate change science is useful in my work.	-3	-2	-1	0	1	2	3	x
Using climate change science in land management is consistent with the mission and objectives of my organization/agency.	-3	-2	-1	0	1	2	3	x
Using climate change science is within my job description and responsibilities.	-3	-2	-1	0	1	2	3	x
Other people in my organization/agency are currently using climate change science.	-3	-2	-1	0	1	2	3	x
Climate change science is useful in long-term land use planning.	-3	-2	-1	0	1	2	3	x
Climate change science is useful for specific management projects.	-3	-2	-1	0	1	2	3	x
Funding is a constraint for addressing climate change in my work.	-3	-2	-1	0	1	2	3	x
Time is a constraint for addressing climate change in my work.	-3	-2	-1	0	1	2	3	x
The politics of climate change are a constraint for using the science in my work.	-3	-2	-1	0	1	2	3	x
I plan to use climate change science in future work that I do.	-3	-2	-1	0	1	2	3	x

Question 2. For this question we are interested in how credible (valid/accurate) you think climate change science is. Please indicate your level of agreement with the following statements:

	Strongly Disagree			Neutral			Strongly Agree	Don't Know
Global and regional climate change science is credible.	-3	-2	-1	0	1	2	3	x
Local (forest stand-level) climate change science is credible.	-3	-2	-1	0	1	2	3	x
Historical data and calculations used in climate change science are credible.	-3	-2	-1	0	1	2	3	x
Projected/modeled future data and calculations used in climate change science are credible.	-3	-2	-1	0	1	2	3	x
I consider science about climate change impacts to be defensible when a decision is challenged or appealed.	-3	-2	-1	0	1	2	3	x
Models that simulate future vegetation scenarios are useful in land management.	-3	-2	-1	0	1	2	3	x
Models that simulate future precipitation patterns are useful in land management.	-3	-2	-1	0	1	2	3	x

Section B: Vulnerability and adaption to climate change impacts

We are interested in your opinions about the likelihood and severity of climate change impacts, and the effectiveness of potential adaptation actions.

Question 3. For this question, think about the impacts of climate change in Idaho and western Montana. For each item please indicate both how LIKELY (column A) you think the climate change impact is, **and** how SEVERE (column B) you think the impact will be.

In the next 20 years, climate change could have these impacts in the Northern Rockies........

	A) How LIKELY is this impact?						B) How SEVERE will the impact be?					
	Very Unlikely		Neither		Very Likely	Don't Know	No Impact		Mod		Very Severe	Don't Know
Increase in mean annual temperatures	-2	-1	0	1	2	x	0	1	2	3	4	x
Changes in seasonal amounts of precipitation	-2	-1	0	1	2	x	0	1	2	3	4	x
Increase in the intensity of precipitation	-2	-1	0	1	2	x	0	1	2	3	4	x

Question 3. Continued….	A) How LIKELY is the impact?						B) How SEVERE will it be?					
	Very Unlikely		Neither		Very Likely	Don't Know	No Impact		Mod		Very Severe	Don't Know
More rain and less snow in winter months	-2	-1	0	1	2	x	0	1	2	3	4	x
Earlier peak streamflow	-2	-1	0	1	2	x	0	1	2	3	4	x
Decrease in total annual streamflow	-2	-1	0	1	2	x	0	1	2	3	4	x
Increase in stream temperatures	-2	-1	0	1	2	x	0	1	2	3	4	x
Changes in where plant species occur on the landscape	-2	-1	0	1	2	x	0	1	2	3	4	x
More wildfire each year	-2	-1	0	1	2	x	0	1	2	3	4	x
Increase in the amount of area burned by wildfire	-2	-1	0	1	2	x	0	1	2	3	4	x
More severe fires	-2	-1	0	1	2	x	0	1	2	3	4	x
More disease and insect outbreaks	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x

Question 4. For this question, think about different ways to adapt and mitigate climate change impacts in Idaho and Western Montana. For each management action please consider whether you think the action would be EFFECTIVE (Column A) for adapting to climate change impacts, **<u>and</u>** how likely is it that the organization/agency you work for WILL TAKE ACTION (Column B) to reduce the potential impacts. Please provide two answers per row.

	A) Would this be EFFECTIVE for adapting to climate change impacts?						B) How LIKELY is your organization/ agency to do this and <u>specifically address climate change impacts</u>?					
	Very Ineffective		Neither		Very Effective	Don't Know	Not at all		Mod		Extremely Likely	Don't Know
Forest treatments to improve forest health	-2	-1	0	1	2	x	0	1	2	3	4	x
Forest treatments to reduce fire hazard	-2	-1	0	1	2	x	0	1	2	3	4	x

Question 4 Continued…..

	A) Would this be EFFECTIVE						B) How LIKELY is your organization/agency to do this?					
	Very Ineffective		Neither		Very Effective	Don't Know	Not at all		Mod		Extremely Likely	Don't Know
Forest treatments to increase water supply	-2	-1	0	1	2	x	0	1	2	3	4	x
Prescribed burning	-2	-1	0	1	2	x	0	1	2	3	4	x
Assisted species migration	-2	-1	0	1	2	x	0	1	2	3	4	x
Road and culvert modifications	-2	-1	0	1	2	x	0	1	2	3	4	x
Consideration of alternative species or plant varieties for restoration	-2	-1	0	1	2	x	0	1	2	3	4	x
Development of restoration objectives beyond the Historical Range of Variability (HRV)	-2	-1	0	1	2	x	0	1	2	3	4	x
Insect and disease control	-2	-1	0	1	2	x	0	1	2	3	4	x
Expanded tree planting to promote reestablishment	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x

Section C: To understand more about our workshop participants, we have a few questions about you. All of your answers are confidential.

7. What is your area of expertise (e.g., position title)?

____

8. How many years have you worked in the Northern Rockies?

Less than 5 years
5 – 15 years
More than 15 years

9. Please indicate the *highest level of education* that you have completed (*check one).*

Less than a high school degree
High school degree or GED
Some college or post high school training
Two year technical or associate degree
Four year college degree (BA/BS)
Advanced degree (MS, JD, MD, Ph.D.)

10. Are you Male or Female?

11. Please check the box that most accurately describes your *political orientation* on the following scale:

Very Liberal	Neither	Very Conservative

Please provide any additional comments here. If you need more space attach a separate piece of paper.

Thank you for helping to improve climate change science, communication, and its usefulness in land management. Please feel free to contact either Jarod Blades or Dr. Hall if you have any concerns or additional comments regarding this survey.

Jarod Blades and Dr. Troy Hall
University of Idaho
P.O. Box 441139
Moscow, ID 83844-1139
Phone: 208-885-7164
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University of Idaho
College of Natural Resources

Appendix I: Post-Workshop Survey

Post-Workshop Opinions about Climate Change Science, Impacts, and Forest Management

Conducted by the Northern Rockies Interdisciplinary Research Team

University of Idaho
College of Natural Resources

Please be assured that all answers provided are confidential. This research has been reviewed and approved by the University of Idaho Institutional Review Board.

To maintain your confidentiality, but allow us to match your pre-workshop survey and post-workshop survey with a unique ID, please enter the **last two letters of your first name, the year of your birth, and the workshop location you attended**.

Last 2 letters of your first name:__________ Year of your birth (YYYY):_________

Workshop location you attended:___________

Section A: Usefulness and Credibility of Climate Change Science
We are interested in your opinions about the usefulness and credibility (accuracy/validity) of climate change science in general, and specifically about the information presented during this workshop. For each of the following questions, please select the answer that most closely reflects your

Question 1. Please indicate your level of agreement with the following statements:

	Strongly Disagree			Neutral			Strongly Agree	Don't Know
Climate change science is useful in my work.	-3	-2	-1	0	1	2	3	x
Climate change science is useful in long-term land use planning.	-3	-2	-1	0	1	2	3	x
Climate change science is useful for specific management projects.	-3	-2	-1	0	1	2	3	x
I plan to use climate change science in future work that I do.	-3	-2	-1	0	1	2	3	x
The global climate change science information is useful for land management (modeling and emission scenario information).	-3	-2	-1	0	1	2	3	x
I plan to use global climate change science in future work that I do.	-3	-2	-1	0	1	2	3	x
The regional climate and water research is useful for land management.	-3	-2	-1	0	1	2	3	x
I plan to use the regional climate and precipitation research in future work that I do.	-3	-2	-1	0	1	2	3	x
The regional vegetation and fire research is useful for land management.	-3	-2	-1	0	1	2	3	x
I plan to use the regional vegetation and fire research in future work that I do.	-3	-2	-1	0	1	2	3	x
The local-scale forest vegetation and climate simulations are useful for land management.	-3	-2	-1	0	1	2	3	x
I plan to use the local-scale forest vegetation and climate simulations in future work that I do.	-3	-2	-1	0	1	2	3	x

Question 2. For this question we are interested in how credible (valid/accurate) you think the climate change science presented at the workshop is. Please indicate your level of agreement with the following statements:

	Strongly Disagree			Neutral			Strongly Agree	Don't Know
Global and regional climate change science is credible.	-3	-2	-1	0	1	2	3	x
Models that simulate future vegetation scenarios are useful in land management.	-3	-2	-1	0	1	2	3	x
Models that simulate future precipitation patterns are useful in land management.	-3	-2	-1	0	1	2	3	x
Historical data and calculations used in climate change science are credible.	-3	-2	-1	0	1	2	3	x
Projected/modeled future data and calculations used in climate change science are credible.	-3	-2	-1	0	1	2	3	x
I consider science presented in the workshops about climate change impacts to be defensible if a decision is challenged or appealed.	-3	-2	-1	0	1	2	3	x

Section B: Vulnerability and adaption to climate change impacts
Based on the information we presented at the workshop, please share your opinions about the likelihood and severity of climate change impacts, and the effectiveness of potential adaptation actions.

Question 3. For this question, think about the impacts of climate change in Idaho and western Montana. For each item please indicate both how LIKELY (column A) you think the climate change impact is, **and** how SEVERE (column B) you think the impact will be.

In the next 20 years, climate change could have these impacts in the Northern Rockies........

	A) How LIKELY is this impact?						B) How SEVERE will the impact be?					
	Very Unlikely		Neither		Very Likely	Don't Know	No Impact		Mod		Very Severe	Don't Know
Increase in mean annual temperatures	-2	-1	0	1	2	x	0	1	2	3	4	x
Changes in seasonal amounts of precipitation	-2	-1	0	1	2	x	0	1	2	3	4	x
Increase in the intensity of precipitation	-2	-1	0	1	2	x	0	1	2	3	4	x

Question 3. Continued….

	A) How LIKELY is the impact?						B) How SEVERE will it be?					
	Very Unlikely		Neither		Very Likely	Don't Know	No Impact		Mod		Very Severe	Don't Know
More rain and less snow in winter months	-2	-1	0	1	2	x	0	1	2	3	4	x
Earlier peak streamflow	-2	-1	0	1	2	x	0	1	2	3	4	x
Decrease in total annual streamflow	-2	-1	0	1	2	x	0	1	2	3	4	x
Increase in stream temperatures	-2	-1	0	1	2	x	0	1	2	3	4	x
Changes in where plant species occur on the landscape	-2	-1	0	1	2	x	0	1	2	3	4	x
More wildfire each year	-2	-1	0	1	2	x	0	1	2	3	4	x
Increase in the amount of area burned by wildfire	-2	-1	0	1	2	x	0	1	2	3	4	x
More severe fires	-2	-1	0	1	2	x	0	1	2	3	4	x
More disease and insect outbreaks	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x

Question 4. For this question, think about different ways to adapt and mitigate climate change impacts in Idaho and Western Montana. For each management action please consider whether you think the action would be EFFECTIVE (Column A) for adapting to climate change impacts, **and** how likely is it that the organization/agency you work for WILL TAKE ACTION (Column B) to reduce the potential impacts. Please provide two answers per row.

	A) Would this be EFFECTIVE for adapting to climate change impacts?						B) How LIKELY is your organization/ agency to do this and specifically address climate change impacts?					
	Very Ineffective		Neither		Very Effective	Don't Know	Not at all		Mod		Extremely Likely	Don't Know
Forest treatments to improve forest health	-2	-1	0	1	2	x	0	1	2	3	4	x
Forest treatments to reduce fire hazard	-2	-1	0	1	2	x	0	1	2	3	4	x

Question 4 Continued…..	A) Would this be EFFECTIVE						B) How LIKELY is your organization/agency to do this?					
	Very Ineffective		Neither		Very Effective	Don't Know	Not at all		Mod		Extremely Likely	Don't Know
Forest treatments to increase water supply	-2	-1	0	1	2	x	0	1	2	3	4	x
Prescribed burning	-2	-1	0	1	2	x	0	1	2	3	4	x
Assisted species migration	-2	-1	0	1	2	x	0	1	2	3	4	x
Road and culvert modifications	-2	-1	0	1	2	x	0	1	2	3	4	x
Consideration of alternative species or plant varieties for restoration	-2	-1	0	1	2	x	0	1	2	3	4	x
Development of restoration objectives beyond the Historical Range of Variability (HRV)	-2	-1	0	1	2	x	0	1	2	3	4	x
Insect and disease control	-2	-1	0	1	2	x	0	1	2	3	4	x
Expanded tree planting to promote reestablishment	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x
Other:	-2	-1	0	1	2	x	0	1	2	3	4	x

Section C: Workshop Evaluation

We are interested in your opinions about how the climate change workshop was conducted in terms of the information presented, and workshop coordination, facilitation, and the processes.

Question 5. Please indicate your level of agreement with the following statements:

	Strongly Disagree			Neutral			Strongly Agree
Scientific information and results were translated for practical use.	-3	-2	-1	0	1	2	3
Information needs were connected with sources of information.	-3	-2	-1	0	1	2	3
The workshop created a forum for individuals who otherwise would not have occasion to work together on these topics.	-3	-2	-1	0	1	2	3

Question 5 continued…..	Strongly Disagree			Neutral			Strongly Agree
The workshop encouraged the use of models and tools for linking science and decision making.	-3	-2	-1	0	1	2	3
Active listening took place during the Q&A and small group sessions.	-3	-2	-1	0	1	2	3
The small group discussions helped me understanding the presented information.	-3	-2	-1	0	1	2	3
Diverse disciplines and interests were <u>not</u> represented at the workshop.	-3	-2	-1	0	1	2	3
The workshop promoted information exchange between scientists, agency and interested stakeholders.	-3	-2	-1	0	1	2	3
The workshop added value by combining data and information from multiple sources.	-3	-2	-1	0	1	2	3
The workshops helped identify the underlying assumptions of the information presented.	-3	-2	-1	0	1	2	3
The workshop helped to understand how research could be used in decisions being made.	-3	-2	-1	0	1	2	3
The workshop was accountable to both resource specialists and decision-maker needs and interests.	-3	-2	-1	0	1	2	3
There was a clear dissemination strategy for workshop information and outcomes.	-3	-2	-1	0	1	2	3
I am confident that information and outcomes from the workshop will be shared with the participants.	-3	-2	-1	0	1	2	3

<u>Learning Environment</u>	Strongly Disagree			Neutral			Strongly Agree
It was easy for participants to speak openly.	-3	-2	-1	0	1	2	3
I was comfortable talking about any concerns or disagreements.	-3	-2	-1	0	1	2	3
Different opinions were welcome.	-3	-2	-1	0	1	2	3
There was adequate time to reflect on new information.	-3	-2	-1	0	1	2	3
The workshop helped participants engage in productive debate.	-3	-2	-1	0	1	2	3

Please provide any additional comments here. If you need more space attach a separate piece of paper.

Thank you for helping to improve climate change science, communication, and its usefulness in land management. Please feel free to contact either Jarod Blades or Dr. Hall if you have any concerns or additional comments regarding this survey.

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Appendix K: Example Interview Questions, Themes, and Participant Quotations for High Level Codes

USEFULNESS

Example interview questions

- Do you use climate change science in the work that you do? How?
- What makes [climate change science] useful or impedes its usefulness [in the work you do]?

Common response themes

- Cursory language about climate change science is used in regional land-use planning documents, environmental impact statements (EISs), and environmental assessments (EAs)
- Scale is an issue; climate change science is not local or site-specific enough to be useful

Example participant quotes

- "Generally, we say the link between greenhouse gas emissions and climate change should be discussed, the capacity of a project to adapt to projected climate change effects disclosed, if there are going to be significant emissions, the cumulative emissions, recognizing that it's a global cumulative effect issue."
- "The projects that I work on [require] very site-specific analysis. Trying to use the current [climate change] research, which is global in many cases or national, and trying to bring that down to the site-specific level and use it meaningfully in project analysis... there just isn't any way right now."

RESPONSE EFFICACY

Example interview questions

- Are you aware of forest management actions that could reduce climate change impacts?

Common response themes

- Familiar management actions that meet multiple objectives are more likely to be used to adapt to climate change
- Increasing "resilience" will increase capacity of ecosystems to adapt to climate change
- Management focuses on restoring ecosystems to reference conditions using the historic range of variability (HRV) concept; restored ecosystems will be better able to adapt

Example participant quotes

- "Management activities that reduce [tree] density [and] improve resilience to fire and drought are going to be consistent with management activities [to reduce climate change impacts]."
- "[The best management option is to] have a diversity of age classes and species represented on the landscape... [so] there's something out there that will be resilient in the future."

- "The thinking [in the USFS]...has been that if we restore things to within the historical range of variability, we somehow increase resistance and resilience to change."

COLLECTIVE EFFICACY

Example interview questions

- How confident do you feel in the ability of **your organization or agency** to take actions to reduce the potential impacts of climate change? Will they do it?

Common response themes

- Institutional barriers mean that managers are unable to treat enough land to effectively adapt to climate change

Example participant quotes

- "There are social barriers that... limit our ability to manage down to such a small fraction of [the] overall landscape that I don't think we're going to [get] to a point that we can [have] any measurable or significant effect."

BARRIERS

Example interview questions

- Are there organizational barriers that impede [the] usefulness [of climate change science]?

Common response themes

- Time, funding, and politics (esp. concerning litigation and public perceptions)
- Informational barriers such as accessibility and applicability limit usefulness
- A lack of organizational capacity, esp. training and/or education about climate change and potential management actions to respond to it
- Inconsistent direction (from line officers, etc.) means climate change is not prioritized in planning efforts
- Bureaucracy makes the process of getting any new ideas/actions implemented incredibly slow

Example participant quotes

- "Without extra resources in terms of capacity or funding, how are [resource managers] supposed to do [anything about climate change]."
- "The hardest thing is having the time to know all the latest, greatest science that's out there, and to have it readily available at your fingertips. We just don't have time to sit there and read everything."
- "[The USFS] still has an education job to do, particularly with folks on the forests and ranger districts, who are out there making these projects go, just to get them... tuned into considering [climate change]."
- "We're still kind of waiting for more of that top-down type of direction in terms of how we're supposed to consider and incorporate climate change into our forest planning efforts and effects analysis for projects."
- "...the Forest Service does not have a history of reacting to change very quickly."