AN ABSTRACT OF THE DISSERTATION OF

Emily Julia Comfort for the degree of Doctor of Philosophy in Forest Resources presented on December 2, 2013.

Title: Trade-offs Between Management for Fire Risk Reduction and Northern Spotted Owl Habitat Protection in the Dry Conifer Forests of Southern Oregon.

Abstract approved:

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There is a perceived trade-off between fire risk reduction and northern spotted owl habitat protection in dry-conifer forests in southwestern Oregon. Management options for balancing this trade-off need to be sought at the landscape level. Applied landscape ecology suggests three important features to consider are (1) patch size and configuration of fire resilient features that are used by northern spotted owls, (2) the scale at which forest structure and configuration impacts fire and northern spotted owl use, and (3) landscape memory, the influence of past ecological process on current ones. In this dissertation, I examine the landscape ecology of past and current fire regimes in the southwestern Oregon and northern California Klamath region and there implications for northern spotted owl habitat and fire risk reduction in current landscapes.

Current old-forests in southwestern Oregon developed under a different fire regime than we see today. In chapter one, I review and discuss the characteristics of past fire regimes both at a fine scale, as assessed by stand-level fire history studies, and at coarse scales, as assessed by lake sediment core studies. Current landscape composition and structure represents a departure from pre-Euro-American landscapes. Forested landscapes generally have higher densities and more homogenous species composition today.

In chapter two, I look specifically at edges within landscapes. Whereas past landscape edges were defined by disturbance patterns/gradients and physiographic changes, current landscapes are largely defined by ownership and management

boundaries. I found no evidence of differences in surface fuel structure between two structurally and compositionally different edge-types, but I did find that edge-type impacted disturbance severity following the 2002 Timbered Rock Fire. On private industry land, salvage-logging was conducted immediately following the fire and resulted in strong edge effects into the adjacent publicly managed land that extended over 250 m from ownership boundaries. Additionally, I found that large gradients in forest age-structure on public lands reduced fire severity.

In chapter three, I examine spotted owl habitat selection of fire-created edges following the 2002 Timbered Rock fire. Spotted owl use of edges varied by edge type (diffuse and hard) and spatial scale. Over moderate to large spatial scales, spotted owls selected diffuse edges, which were characterized by shallow gradients in fire severity. At fine spatial scales (<0.8 ha), there was some evidence of an association between spotted owls and hard edges, which were characterized by very steep gradients in fire severity. However, at broader spatial scales this result was reversed.

I finish with overall conclusions that identify areas of common ground across these disciplines and lines and evidence, and with some management recommendations to restore more historic landscape structure that may assist with both fire risk reduction and northern spotted owl conservation.

Trade-offs Between Management for Fire Risk Reduction and Northern Spotted Owl Habitat Protection in the Dry Conifer Forests of Southern Oregon

by
Emily Julia Comfort

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I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my dissertation to any reader upon request.

Emily Julia Comfort, Author

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CHAPTER 1: LANDSCAPE FIRE ECOLOGY IN THE KLAMATH-SISKIYOU REGION

Abstract

The characteristics of past fire regimes can help us understand the patterns and processes that shaped the current landscape and desired future landscaped. In this chapter, I review both fine- and coarse- scale fire regimes in the Klamath-Siskiyou region of southern Oregon and northern California. Fire regime characteristics such as fire frequency, fire size, and fire severity varied considerably with physiographic conditions, but were generally frequent (7-42 years median fire return intervals) and small (<150 ha) and mixed-severity. The variability of these characteristics, including variability in fire free intervals, however, probably had a great impact on the high diversity of plant and animal species observed in the region. Current landscape composition and structure represents a departure from pre-Euro-American landscapes. Forested landscapes generally have higher densities and more homogenous species composition today. This change in pattern can have a large impact on fire effects and may change the way that fire interacts with the landscape. This could be exacerbated by future climate change. Because fire has been one of the drivers of biodiversity in the past, these changes will have consequences for plant and animal dispersal in response to climate changes. While management practices that are based strictly on past fire patterns may not be feasible, we may be able to incorporate these principles into a multi-scaled management approach that can facilitate desired future conditions.

Introduction

The dry-mixed conifer forests that are common on federally-managed land in southwestern Oregon are generally classified as a mixed-severity fire regime, the drivers and features of which are the subject of several reviews (Agee 2005, Halofsky et al. 2011, Perry et al. 2011) and ongoing scientific debate (Hanson et al. 2009, Spies et al. 2010, Williams and Baker 2012, Fule et al. *In press*). In this review, I will summarize research that has quantified characteristics of fire in the Klamath-Siskiyou region and its neighborhood, including research I conducted outside of my dissertation. Fire regimes have also been regulated by people throughout time. I discuss the changes in our interactions with landscape since Euro-American settlement. Applied historical ecology (Swetnam et al. 1999) is a useful tool for managers, but must be combined with current ecological context (resource needs of humans, wildlife, and plants) and future changes to the drivers and controls of ecological processes. I conclude with a discussion of how variability and complexity of past fire can be incorporated in current management policies.

Fire regimes vary in both space and time. At small spatial extents, they are generally determined by physiographic controls, fuel accumulation, and weather events (Heyerdahl et al. 2001, Taylor and Skinner 2003). At large spatial extents, fire regimes are related to climate variability (Whitlock et al. 2003, Littell et al. 2009, Trouet et al. 2010). Over millennia, plants and wildlife migrate in response to changing climate and disturbance regimes (Whitlock 1992). The multi-scaled characteristics of fire history are important to understanding the drivers and impacts of fire on forest composition and wildlife habitat. Post-Euro-American-settlement fire suppression, timber harvest, grazing, and land conversion has erased or obscured the evidence of past community structure and composition and altered their development patterns. Additionally, areas that have been managed in the past century are not a random selection of what was available, but purposely selected from what was available a century and a half ago and more recently depending on the technology available.

Dry mixed-conifer forests are variable in both composition and ecology (Franklin and Dyrness 1988, Hessburg et al. 2005, Stephens and Moghaddas 2005). In southwestern Oregon, there is a convergence of three ecoregions: the Klamath, Southern Cascades, and the Oregon Coast Range which leads to a variety of mixed-conifer-type associations in addition to mixed-evergreen stands, oak woodland and savannas, and more open shrub fields and grasslands. Mixed severity fire regimes are highly variable and not well defined (Taylor and Skinner 1998, Halofsky et al. 2011). In the Klamath- Siskiyou region, the diverse flora and fauna are credited to the assortment of forest composition and structure created in part by the history of mixed severity fire on the landscape. In his exhaustive monograph on the vegetation of the Siskiyou mountains, Whittaker (1960) concluded that the dry mixed-conifer forest is a fire climax system in which the stable vegetation is maintained by population instability caused by frequent fire disturbances. He attributed the great floristic diversity to the extreme variation in climate and parent material created by the age and complexity of the Siskiyous and the surrounding mountain ranges and the prevalence of fire. Even within the diversity of flora, the variation in climate from the west side of the Siskiyous to the eastern side creates different vegetation communities (Waring 1969).

Fire Regimes

Fire regimes are the generalized pattern of fire occurrence for a defined region over a defined period of time (Agee 1996). Predictable patterns are typically described by quantifiable traits like frequency, the number of fires in a given area over a given time frame, and severity, the effects of fire on ecological structures (typically the relative amount of surviving vegetation). Fire history studies help to describe fine-scale patterns from the recent past by examining evidence from fire scars on live trees, stumps, snags, and logs. This information can be combined with stand development patterns determined from tree ring evidence to estimate fire frequency and in some cases measures of severity and fire extent for the current stands of trees. Coarse-scale fire history studies use evidence from lake sediment cores and background charcoal deposit levels as well as spikes in charcoal deposition to estimate characteristics of fires over lake basins (and beyond) over millennia. The level of background charcoal deposit can be an indication of the amount of biomass available for burning under frequent fire return intervals (Whitlock and Millspaugh 1996, Whitlock et al. 2004). Peaks in charcoal deposition can indicate periods of unusual fire activity. This can be severe fires near to the lake or regional fire events (Long et al. 1998). Taken together, evidence from these two lines of study can provide valuable information about the role of fire in past and present and how we can expect fire to influence future forest ecosystems.

Frequency

Fire frequency in the Klamath-Siskiyou region is unique because of its extreme variability (Taylor and Skinner 1998). While the region shares dominant species composition with other regions, it has experienced both frequent fires in the past and long fire- free periods, which requires unique adaptations for shorter-lived associates and results in unique and varied stand structure and landscape pattern. This also complicates the issue of how far the current landscape composition and structure has departed from its normal range of variation.

In the recent past, fire frequency has been variable over both time and space. Beaty and Taylor (2001) found median composite fire return intervals (MFRIs) varied by slope aspect, slope position, and elevation in the nearby Southern Cascades (Table 1.1). The longest MFRIs and fire rotations were on north-facing, higher elevation slopes and the shortest were on south-facing, lower elevation slopes. While MFRI were slightly shorter during the early-settlement period than the pre-settlement period, they increased during the fire suppression era. Similarly, Taylor and Skinner found that MFRIs were longer on north-facing slopes (2003) and north and east-facing slopes (1998) at two study areas in the Klamath Mountains, but they did not seem to vary by species composition or elevation. At elevation less than 1000 m fire tends to be more frequent, but have similar variability (Wills 1991, Comfort et al. *In Preparation*). Mean return intervals for larger-scale fire events (fires that burned multiple plots or larger areas within studies) were longer, but still appear to increase with elevation (Table 1.2). In total, 70 larger-scale fires were recorded between 1630 and 1987. Seven years were identified as regional fire episodes (larger-scale fire events in two or more studies): 1738, 1742, 1795, 1838, 1848, 1858, and 1987.

Over longer time scales, fire episodes (decades-long periods of high fire activity) occurred between 3 and 17 times per 1,000 years in the upper elevations of the Klamath Mountains, varying with large-scale climate changes since the last glacial period (Mohr et al. 2000, Whitlock et al. 2004, Briles et al. 2005, Briles et al. 2008) (Table 1.3). Frequency of fire episodes appears to be greater for high-elevation sites than for lower elevation.

Fire Severity

Fire severity is driven by both bottom-up factors (fuel properties) and top-down factors (weather events, climate trends). While the same factors have influenced fire severity in the past, the current structure and composition of the landscape may interact with fire differently than prior to Euro-American settlement.

Beaty and Taylor (2001) found that in the last 350 years, fire severity varied mostly by slope position in the Southern Cascades. Lower slope positions had lower-severity fire. Middle slope positions had mixed, moderate- and low-severity fire. Upper slope positions had the highest-severity fire. Taylor and Skinner (1998) found that fire severity was mostly low (59%) in a 1570 ha study area in the Klamath mountains. High- and moderate-severity fire (fewer than 20 residual trees/ ha) were generally found in upper slope positions in the study area. In two recent fires, the Quartz fire (2500 ha in 2001) and the Big Bar fire (50,590 ha in 1999), fire severity overall was mostly low (17% and 76% respectively) and moderate (54% and 8% respectively). High-severity fire only accounted for between 16-29% of the total fire area (Alexander et al. 2006).

In recent fires, fire severity has been associated with pre-fire vegetation structure and age (Odion et al. 2004, Thompson et al. 2007, Thompson and Spies 2010, Thompson et al. 2011), aspect (Alexander et al. 2006), and extreme fire weather (Thompson and Spies 2009). Odion et al. (2004) examined a 500,000 ha region in the

Klamath National Forest and determined that fire severity in a 1987 fire remained low in long unburned areas that are currently dominated by closed canopy forests. In contrast, Miller et al. (2012) found a non-linear relationship between fire severity and time since fire that differed among forest types. In Douglas-fir forests, the proportion of high-severity fire in wildfires that occurred after 1987 was similar between long unburned areas (no fire since 1910) and areas that had experienced a prior fire within 30 years (9% and 10% respectively), however areas that burned 31 to 98 years prior had significantly less high-severity fire (5%). In mixed-conifer forests, there was a reduction in the amount of high severity fire for any area that had burned in the last 98 years compared to long unburned areas (12% high severity for 1-60 years, 13% for 61-98 years, 16% for long unburned). Miller et al. (2012) also found that fire severity was lower for forests with large conifers than for forest with small diameter trees and hardwoods.

Fire severity has self-reinforcing properties. Unmanaged areas in the 2002 Biscuit fire that were within the fire boundary of the 1987 Silver fire tended to reburn at similar severity (Thompson et al. 2007, Thompson and Spies 2010), however, stands that had been salvage-logged following the Silver fire tended to burn at high severity. Severity in the Biscuit fire was also associated with extreme fire weather (Thompson and Spies 2009).

Extent/ Patch size

Under a mixed-severity fire regime, spatial extent of fire would be expected to vary in space and time according to both topographic controls (barriers, changes in fuel structure) and climate controls. Under cooler, wetter conditions, fire extent would be limited by fuel moisture and during period of hot, dry conditions, fire extent would be limited by ignitions and fuel continuity.

Beaty and Taylor (2001) found that average extent of fires from 1704 to 1926 was approximately 106 ha, but that extent was highly variable from year to year. The extent also varied according to aspect, potential moisture, and species composition. Fires of all sizes were generally more common during dry years and large fires were more common when dry conditions over a three-year period were preceded by wet conditions (Beaty and Taylor 2008). These findings are consistent with the interactions we would expect between weather and fuels. During dry years, weather conditions will drive fires where topography and fuels are conducive. In dry years following wet years, vegetation has recently had an increase in growth more uniformly across the landscape, so fuel accumulation are likely to be higher and more continuous than in years preceded by dry years. This can lead to more widespread fires.

Taylor and Skinner (2003) found that median fire size was 128 ha (range 25-1541 ha) in the pre-settlement era (prior to 1849) in a 2325 ha study site in the Shasta-Trinity National Forest. In the early-settlement era (1850 to 1904), median fire size was 106 ha (range 25-1188 ha). In the fire-suppression era (1905 to 1995), median fire size was 25 ha. Thirteen fires burned more than 500 ha over the course of their fire history. At a nearby 1570 ha study area in the Klamath National Forest, the authors (1998) report similar findings. They found that fire extent was unrelated to forest species composition, but the average total area of fire within the study area over the course of the study (1626-1987) was 350 ha (+/- 217 ha, 22% of the study area). In 17 individual years, over 500 ha within the study area burned.

While Duren et al. (2012), found that the relative proportion of open landscapes (prairies, shrublands, savannas, and mixed herbaceous/ shrubs) and forested landscapes (greater than 25% cover of mixed conifers and/or hardwoods) remained the same between the early settlement period (late 1850s) and 2005 in a 300,000 ha study area in middle and lower elevation foothills and valleys of the Applegate, Illinois, and Rogue River watersheds, there was some conversion back and forth between individual open and forested sites. Fire was linked to some of these conversions, but not enough to consider it a significant driver of overall landscape change. However, because all forested landscape with greater than 25% tree cover were grouped together, this study did not account for conversions from more open-type forests (savannas and low density woodlands). It may not have captured significant changes in forest structure.

Seasonality

Historically, fires occurred primarily during the dormant season. Fry and Stephens (2006) found that more than 50% of historic fires in mixed-conifer stands in the southern Klamath Mountains occurred during fall and winter months (latewood and outside of the growing season). Beaty and Taylor (2001) found that seasonality varied by slope aspect in the Southern Cascades. On north slopes, fires consistently burned only during the dormant season. While most fires on south-facing slopes also occurred during the dormant season, 13% occurred in the middle or late growing season. Taylor and Skinner (2003) found that 76.2% of fires in their study area in the Klamath Mountains occurred in mid-summer to fall (outside of the growing season). Most fires that occurred during the growing season were recorded in latewood. Less than 7% of fires were recorded in early wood.

Other common forest types

Many of the fire history studies from the Klamath- Siskiyou region included in this review were conducted at higher elevation conifer forests, but other forest types are common at lower elevations and are very important for management consideration, given their importance for as habitat for some wildlife species and overall landscape heterogeneity.

Ponderosa pine woodlands and savannas

Fry and Stephens (2006) examined fire history in 120 ha of the Whiskeytown Natural Recreation Area (Klamath Mountains) dominated by ponderosa pine. They found shorter point and composite MFRIs between 1750 and 2002. Fires that scarred at least 25% of recording trees occurred every three years (range of FRI 1-59). Fires occurred during the late growing season or the dormant season. Synchrony of fire scar occurrence across plots increase after 1850. Fire was rare after 1925 until recent prescribed fire.

In the absence of frequent fire in pine woodland and savannas, tree densities increase and species composition of regenerating trees switch to more shade tolerant conifers and hardwoods. Leonzo and Keyes (2010) looked at the growth rates of relict trees in old stands compared to encroachment trees (<67 years old). They found that relict trees were generally ponderosa pine and sugar pine (76%) and that encroachment trees are predominantly white fir (29%) with large components of Douglas-fir (17%), pines (17%), black oak and canyon live oak (17%), and tanoak (18%). The periodic annual basal area growth rate over ten year periods between the initiation of the encroachment cohort in the 1950s and 2005 remained constant for relict trees and steadily increased for encroachment trees. The authors suggest that when site capacity is reached, the growth and health of relict trees will be impacted by the encroachment cohort.

Oregon white oak and California black oak savannas and woodlands

Oak regeneration in the Klamath- Siskiyou region appears to have been episodic over the last 350 years (Gilligan and Muir 2011, Comfort et al. *In Preparation*). Oak have long been a component of lower elevation landscapes (<1200 m). The last major pulse of establishment was in the mid to late 1800s. Recruitment has been sparse in the last century and many new stems remain small in diameter. The median age of trees under 10 cm dbh was 83 years in one study (Gilligan and Muir 2011) and 71 years for trees under 8 cm dbh in another (Comfort et al., unpublished data).

Encroachment in oak savannas and woodlands by more shade tolerant conifers (Cocking et al. 2012) is a conservation concern due to the importance of oak habitats to bird and other wildlife habitats in southern Oregon (Altman 2011). Hosten et al. (2006) reviewed oak ecology in the Pacific Northwest. They identified fire as an essential tool to oak restoration because it removes competing woody vegetation and restores native herbaceous communities. The authors, however cautioned that oak restoration must be contingent on landscape context. For example, oaks are a transient piece of chaparral ecosystems and should not be a restoration focus in areas that are dominated by chaparral.

Chaparral Shrubs associations

Southern Oregon chaparral systems are compositionally similar, but thought to be fundamentally different in age-structure and fire history than their southern California counterparts. Duren and Muir (2010) examined chaparral age-structure in areas with known fire histories in southwestern Oregon. They found that there was moderate to high survival of shrubs following wildfire and considerable recruitment of new shrubs in the absence of fire suggesting that southern Oregon chaparral fires are highly complex. Shrub communities appear adapted to a highly variable fire return interval consistent with a mixed-severity fire regime.

Riparian

Limited research suggests that fire in riparian areas is similar to the surrounding upland areas (Halofsky and Hibbs 2008). In different fire events, riparian areas can act as both fire barriers and fire corridors. Riparian areas had longer median fire return intervals, but similar ranges in fire return intervals than adjacent upland areas suggesting a more variable fire occurrence in the Klamath Mountains (Skinner 2003) and in the Oregon Southern Cascades (Olson and Agee 2005). In mixed-conifer sites in the Klamath-Siskiyou region, pre-settlement composition was similar in both uplands and riparian areas, while post-settlement, species composition shifted towards fire intolerant species and growth rates of Douglas-fir appeared to decline (Messier et al. 2012). In the interior valley areas, pre-settlement composition of riparian areas was mostly open hardwood savanna and woodlands, suggestive of a high frequency, low-severity fire regime. Post-settlement, conifer survival increased, stem densities increased, and forest canopies closed (Messier et al. 2012). Currently riparian areas are treated as reserves to protect water quality and habitat (USDA 1994).

Human interactions

There is considerable evidence to support the frequent use of fire by Native Americans prior to Euro-American settlement for a variety of reasons (LaLande 1995, LaLande and Pullen 1999), however the pattern of those fires is not recorded (Atzet and Wheeler 1982). Results from many of the studies I reviewed suggest that during the early settlement period (prior to 1900), fire may have been slightly more frequent than it had been prior to euro-American settlement and fire frequency declines abruptly in the early 1900s (Taylor and Skinner 1998, Beaty and Taylor 2001, Taylor and Skinner 2003, Comfort et al. *In Preparation*). Reports of early settlers using fire to clear brush and improve access to mines, ease travel, and open up rangelands (Ringland 1916) support this finding. In 1910, fire suppression became policy for federally managed lands. It was probably not until after World War II, when a smoke jumper base was established in Cave Junction, that fire suppression was effective. These changes in the use of fire by humans in addition to other human interactions with the landscape (selective logging (Naficy et al. 2010) and grazing (not well documented, but discussed in Huago (2010) , Duren and Muir (2010), and Gillian and Muir (2011)) have altered stand-scale heterogeneity and structure.

In lower elevations of the interior valleys of the Siskiyou Mountains, fire acted as a mortality agent that reduced recruitment of new trees prior to Euro-American settlement. Following settlement, fire became a regeneration agent that opens up the forest canopy and allows recruitment of new cohorts of trees (Comfort et al. *In Preparation*). Comfort et al. (*In Preparation*) reconstructed stand development history at 119 plots systematically located across the 34,000 ha Middle Applegate watershed using a dendrochronology approach. They found that currently, 98 out of 119 plots had densities greater than 200 trees over 8 cm diameter at 1.4 m (dbh) per hectare (114 have at least one tree). The average tree density for all plots (including open plots) was 667 trees > 8 cm dbh per hectare. A reconstruction of landscape structure from the same location using General Land Office (GLO) survey from the late 1800s suggests that forest density was already 272 trees per hectare across the study area and that much of the area was forested (Baker 2011). Comfort et al. (*In Preparation*) showed a lower overall tree density across the Middle Applegate watershed in 1900 (168 TPH), but also found that only 44 out of 119 plots currently have more than 200 TPH that established prior to 1900 (103 plots out of 119 had at least one tree that established prior to 1900). Only six plots had at least one tree that established prior to 1750 (Comfort et al., unpublished data). Over a century and a half, the landscape has transformed from a fire-maintained, open landscape with scattered trees and some pockets of woodlands to a mostly dense forest where conditions under which fire could not be suppressed would likely lead to fire that kills many overstory trees.

Species composition has also shifted in the last 200 years at these lower elevation, interior valley sites. Baker's GLO reconstruction (2011) from the Middle Applegate watershed identified four community types. Two were 20 to 52% Douglas-fir with minor components of either madrone or madrone, oak, and pine. Two were less than 20% Douglas-fir and were dominated by oaks or oaks and pines (Baker 2011). Comfort et al. (*In Preparation*) found that composition was strongly related to elevation and heat load (a measure of heat input based on latitude, slope, and folded aspect (McCune and Keon 2002)). Comfort et al. (*In Preparation*) also found that species composition across the watershed appeared to be shifting toward more shade tolerant species, but the relative shade tolerance varied by physiographic groups. For instance, at lower elevations and high heat loads, plots were largely composed of Oregon white oak and California black oak in 1875, but by 2000, the two oak species represent less than 1/3 of the total number of stems (Figure 1.1). The changes were more subtle at high elevation, low heat load sites. Douglas-fir has long dominated these sites, but the proportions of white fir, canyon live oak, and black oak have increased consistently over time (Figure 1.1).

In higher elevation forests changes are also being observed. Once patchy distributions of fire-created openings are disappearing. Skinner (1995) found that forest openings had decreased in area and increased in number between 1944 and 1985 based on aerial photographs from the Klamath Mountains. The author attributed it to encroachment of islands within patches and narrowing of the base of fingers off of older, large openings that cut off small openings from one another. Only three of fifty sites had newly created openings between 1944 and 1985. One was attributed to a tree fall gap and the other two to mudslides. Sensenig et al. (2013) found that patterns of within-stand development, particularly early growth rates, were significantly different between old growth trees and trees that established following severe disturbances in the late 1800s and early 1900s. Young growth trees had rapid early annual growth (first 10-20 years) followed by steadily declining annual growth. Old-growth trees had lower early annual growth, but annual growth increase during the first 50 years before declining.

Compositional changes are also apparent between old growth trees (>100 years) and younger cohorts (Taylor and Skinner 2003) at higher elevation forests. In the Hayfork study area in the Shasta-Trinity National Forest, old forests are primarily composed of fire-tolerant and shade-intolerant ponderosa and sugar pine. Young trees (<100 years) are generally more fire-sensitive and shade-tolerant species, primarily Douglas-fir, white fir, and incense cedar (Figure 1.2).

Human interactions have also altered fire effects at landscape-scales considerably. Ownership and management units now drive overall landscape pattern of age class and species composition. Within-ownership boundaries intensive management (Omi and Kalabokidis 1991, Thompson et al. 2011), salvage logging (Donato et al. 2006, Thompson et al. 2007), and road creation (Narayanaraj and Wimberly 2013) impact fire behavior as well as species composition. Patch sizes of different ages are likely very different today than in the past. A study that compared early 1900s landscape composition (1932 to 1966) to more recent landscape composition (1981 to 1993), found that the upper Klamath region today has less 22%

less young, multi-story forest and 12% less old forest (Hessburg et al. 2000). Wildfires in the Pacific Northwest are increasing in size and frequency due to longer and drier fire seasons (Westerling et al. 2006) and this trend may continue with future climate change (Westerling and Bryant 2008). Some, however, suggest that we are in a fire deficit and need more fire to create unmanaged early seral habitat (Hanson et al. 2009). These changes have implication not just for fire behavior, but also for plants and animals that are adapted to natural disturbance regimes.

At both stand- and landscape- scales, southwestern Oregon forests have changed since Euro-American settlement. The degree to which these changes are within the bounds of past variation at larger temporal-scales is difficult to determine.

Management Concerns

Currently, land managers in the dry-forest regions of the Klamath-Siskiyou region are concerned that there is high risk of large, high-severity fire and other density-dependent disturbances (like insect outbreaks). This concern is supported by both climate evidence and fuel evidence. Climate is a strong driver of fire regimes both in the recent past, regionally (Whitlock 2001, Heyerdahl et al. 2002, Whitlock et al. 2003, Marlon et al. 2009) and over large spatial scales (Littell et al. 2009, Trouet et al. 2009, Trouet et al. 2010). Regional models that examine changes in fire risk under climate change, suggest that, in addition to increases in fire size and frequency (Westerling et al. 2006, Westerling and Bryant 2008), controls on fire regime may shift from top-down (climate) control to bottom-up (fuel) controls (Littell et al. 2009). Dry-forests are locally at risk because there are few old, fire-resistant trees and forests are dense with multiple canopy layers, continuous canopies, and higher fuel loads and fuel continuity (Franklin and Johnson 2012).

This high risk of fire can have impacts on many forest management concerns such as wildland firefighter safety, protection of northern spotted owl habitat, prevention of fire spread to the wildland-urban interface, and protection of timber reserves on both public and adjacent private land. Managers are also restricted in their management options because of legislated mandates and policies for land-use (e.g. The Northwest Forest Plan, the Endangered Species Act, the O&C lands Act). Disagreements over historic conditions and current objectives have reduced the management options on federal lands. Currently, fuels reduction treatments near the wildland-urban interface are common, but larger-scale management projects are rare and controversial. There is a short-term, risk-averse paradigm that may be increasing the long-term ability of plants and animals to adapt to future climate changes and disturbance regimes (Maguire and Albright 2005). Attempts have been made to model long- and short- term risk trade-offs for northern spotted owls (Roloff et al. 2005, Ager et al. 2007, Roloff et al. 2012) and fish (O'Laughlin 2005), but models have a lot of uncertainty associated with them. An ecological forestry demonstration project is also underway to engage the public and industry toward a dry-forest management strategy (Franklin and Johnson 2012) that is geared at reducing fire risk, protecting and enhancing northern spotted owl habitat, and producing timber revenue for counties (Reilly 2012). Monitoring the ability of this management strategy to meet such diverse and apparently conflicting management goals could fill many gaps in our knowledge on management trade-offs.

Multi-scale management targets

In order to emulate past disturbance regimes under an ecological forestry model of management, the multi-scaled character of the fire regime must be considered in addition to the physiological limitations of the desired future vegetation composition and structure. We should set goals at landscape scales, but use local knowledge to determine where on a landscape and within a stand, each goal is best met.

A classic view of vegetation communities classifies them as static associations that are based on temporal and spatial variations of more deterministic "climax" communities (Clements 1936). Others have suggested that vegetation communities are more individualistic and depend on random processes and migration of different species over time (Gleason 1939). Williams (2001) suggested that communities are in equilibrium with climate changes at the scale of thousands of years which is supported by many of the long term fire and vegetation studies reviewed in this paper (Whitlock 2001, Heyerdahl et al. 2002, Whitlock et al. 2003, Marlon et al. 2009). Currently in the Klamath-Siskiyou region, the landscape structure may not be capable of facilitating long-term community equilibrium because small-scale drivers, like disturbance, that affect plant and wildlife functional responses are disconnected. While long fire-free intervals are historically common in the Klamath-Siskiyou region, the landscape context is novel. Large portions of the forested landscape are at the high end of their historic range of variability, there are more barriers to migration (roads, developed land), and there is already a deficit of old-forest.

Variation in the frequency and severity of disturbance are the drivers of both stability and change over course scales (both thousands of years and miles). In the past climate variation has driven shifts in vegetation communities, but the success of those shifts has been dependent on a heterogeneous landscape created by multiple scales of disturbance. Given current management constraints and goals and landscape context, how can we use the extensive knowledge of past fire occurrence in this region to increase the resilience of landscape for multiple objectives (e.g. wildlife habitat, and sustainable timber yields)? Variability and heterogeneity are objectives that are difficult to define and manage. Additionally, management is generally prescribed at the scale of decades to centuries, while dispersal (especially plant range shifts) and adaptation are long-term concerns.

Landscape-scale

The goals at the landscape-scale should be to restore patch sizes, configuration, and age structure that are consistent with past fire regimes. In theory this would leave the landscape functions that have fostered range shifts in the past to operate in the face of future climate change. Current barriers to this goal are numerous: disagreements about historic patch size, barriers and filters created by development, public distrust of management practices are just a few. However, creating a patch-landscape mosaic has several, bridge-crossing attributes. Using measureable descriptors of landscape patches from landscape ecology would reduce ambiguity and produce quantifiable variation. Parameters such as size distribution, boundary form, perimeter: area ratio, patch orientation, context, contrast, connectivity, richness, evenness, dispersion, predictability (Wiens et al. 1993) are measureable characteristics of patches that relate directly back to process.

Certain of these parameters are already well studied, but many need more applied research. Fire history studies reviewed in this paper, suggest that patches sizes were typically less than 130 ha, but less commonly were 500 ha or larger in mid- to high-elevation conifer forest. These studies could provide a quantitative foundation for allotting patch size distributions in similar forest types. Continued research that examine the effects of patch attributes directly on process would help further refine their historical and potential future relationship with the current landscape. For example, in chapters three and four of this dissertation, I examine the roles of patch edges in fuel structure, fire effects (fire process), and post-fire use by northern spotted owls (process). These studies suggest that hard patch edges created in mixed-ownership landscapes following fire and salvage-logging (on private land only) do not have different pre-fire fuel structure, but have different fire severity and use by northern spotted owls than very small high-severity fire patches. Similarly, the temporal application of management (harvest or prescribed fire) should match the range of historical fire frequency (i.e. for dry-mixed conifer, frequent application of management to small spatial scales).

Stand-scale

At the stand-level, resilience of important stand structure should be emphasized. Specifically, I mean the stands should be able to absorb disturbance (harvest, fire, or something else), self-organize following disturbance, and adapt to new conditions created by slow changes to the system (Walker et al. 2002). Again, by focusing on retaining structures that increase resilience (or removing structures that decrease resilience) of a process (reduced fire risk, increase wildlife use), the process and required variability is more quantifiable, defendable and useful. In chapter three, my study suggests that northern spotted owls use hard edges only at very small scales (<0.8 ha), so if northern spotted owl use is a desired function of a stand, the maximum tolerable high severity fire or logging patch would be very small.

Tree-scale

In frequent- and-mixed fire regimes, species composition and stand density can be a strong measure of resilience in this sense. Disturbance keeps a variety of age classes and site conditions on the landscape for different plant functional groups to adapt to the new climate conditions. If disturbances are too frequent and severe, early seral species will dominate the landscape and if disturbances are infrequent, late seral species will dominate (Roxburgh et al. 2004). There is evidence to suggest that in lower elevation oak stands and mixed-conifer forest that were once pine dominated, there is both a shift in species composition and an increase in density of less fire tolerant Douglas-fir and fire intolerant Pacific madrone and canyon live oak. Retaining and releasing large and old fire tolerant trees by restoring a less dense condition will increase the ability of these legacy structure to resist future fires. The landscape memory that they provide can influence future stand species composition and genetics.

Conclusions

While I was able to amass a large collection of research directly related to past and current fire regimes in this region, there are still large gaps in our knowledge. A valid criticism of tree- ring based studies of fire history is the lack of account for missing evidence. It is likely that forests had more trees 150 or more years ago than we see evidence of today. That missing data is likely in the form of smaller trees that died and rotted away leaving no trace. This makes it hard to definitively state that densities of regeneration (or even some larger class trees) were not greater than estimated. However, using multiple sources, including density estimates from General Land Office survey reconstructions and modern establishment patterns of different tree species, to better inform interpretations of past patterns lends a better perspective. Additionally, our measures of frequency, severity, and size of fire can only under-estimate the true values because not all trees are scarred by every fire. Many parameters of fire regimes are not easily estimated from historical evidence, so we need to look to current fire events to establish patterns and processes.

The bulk of the research, however, suggests that prior to Euro-American settlement, fires were frequent, common across the landscape, but small in individual size, and of moderate to low severity in mixed conifer forests in the Klamath Siskiyou region. In contrast, most fire ignitions are currently suppressed prior to any significant impact on the land or, alternatively a small portion of ignitions that we are unable to suppress grow very large.

Flexible and adaptive landscape management is particularly important in dry-forests in the Klamath-Siskiyou region. The processes that created the diverse landscape we have today are multi-scaled (Figure 1.3). At small spatial and temporal scales, management should focus on resistance. The susceptibility of legacy individual tree, structures, nest sites should be considered at this scale. At the stand-level and project scale, management should focus on resilience. The loss of resilient, replaceable components of stands, such as shrubs, and small, young trees should not have long-term impacts on future composition and structure, but may have meaningful impact on short-term stand health and retention of irreplaceable stand components.

At the landscape-level, management regimes need to be coordinated and reflect long-term thinking, so the landscape can facilitate adaptation of plants and animals to new climate conditions despite the multiple human-created barriers to this process. This will require thinking about disturbance patch size and connectivity and incorporating land-use on non-public lands. As early as 1900, forest managers were recognizing the important role of fire in shaping southwestern Oregon forests

(Leiberg 1900, Ringland 1916). Leiberg noted that in the all of the 3,000,000 acres he had observed of the Ashland and Cascade Range forest reserves, all but 25,000 ac had signs of past fire. He cited evidence of both recent, human-ignited fires and suggestive evidence (lack of old-growth fire intolerant species and multi-age structure) from long past fires abounded. We need to continue to recognize the both beneficial and destructive nature of fire by thinking of fire regimes as multi-spatial, multi-temporal targets.

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Figures

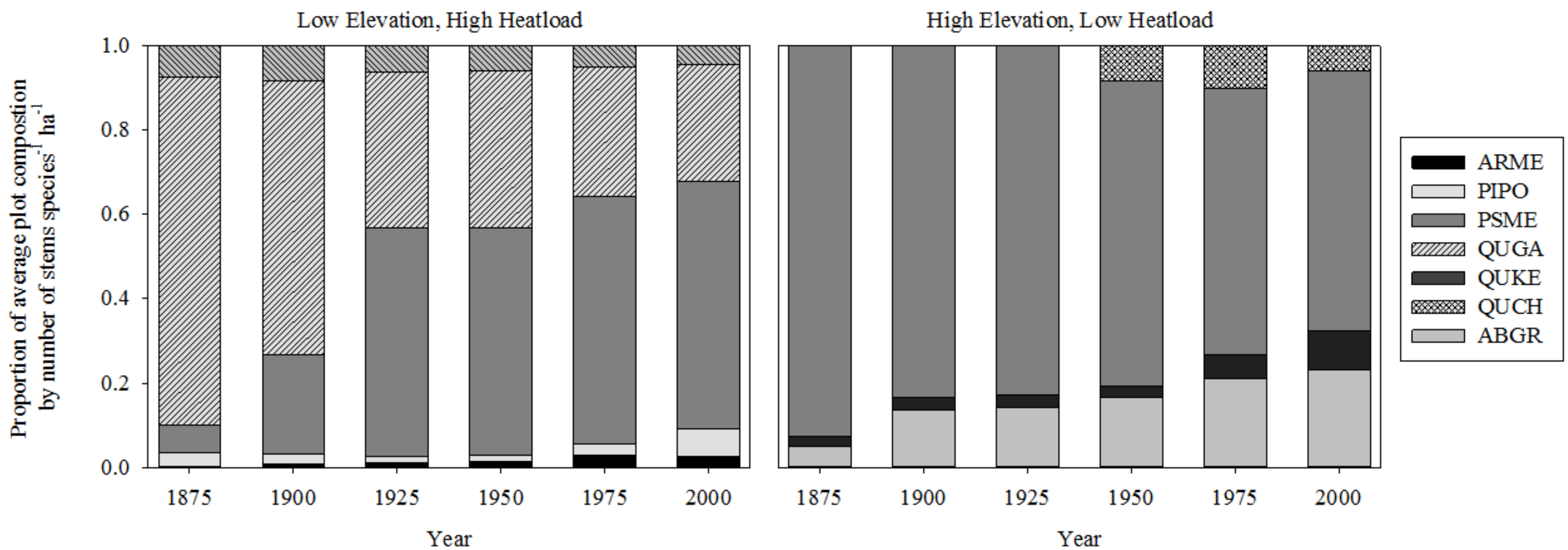

Figure 1.1. Comparison of species composition change over time for two different physiographic groups in the Middle Applegate watershed, OR. Species proportions are the based on the average number of each species per plot (N= 13 plots for low elevation, high heat load and N= 8 for high elevation, low heat load group). ARME = Pacific Madrone; PIPO= ponderosa pine; PSME= Douglas-fir, QUGA= Oregon white oak, QUKE= California black oak, QUCH= canyon live oak, ABGR= white fir (Comfort et al., unpublished data).

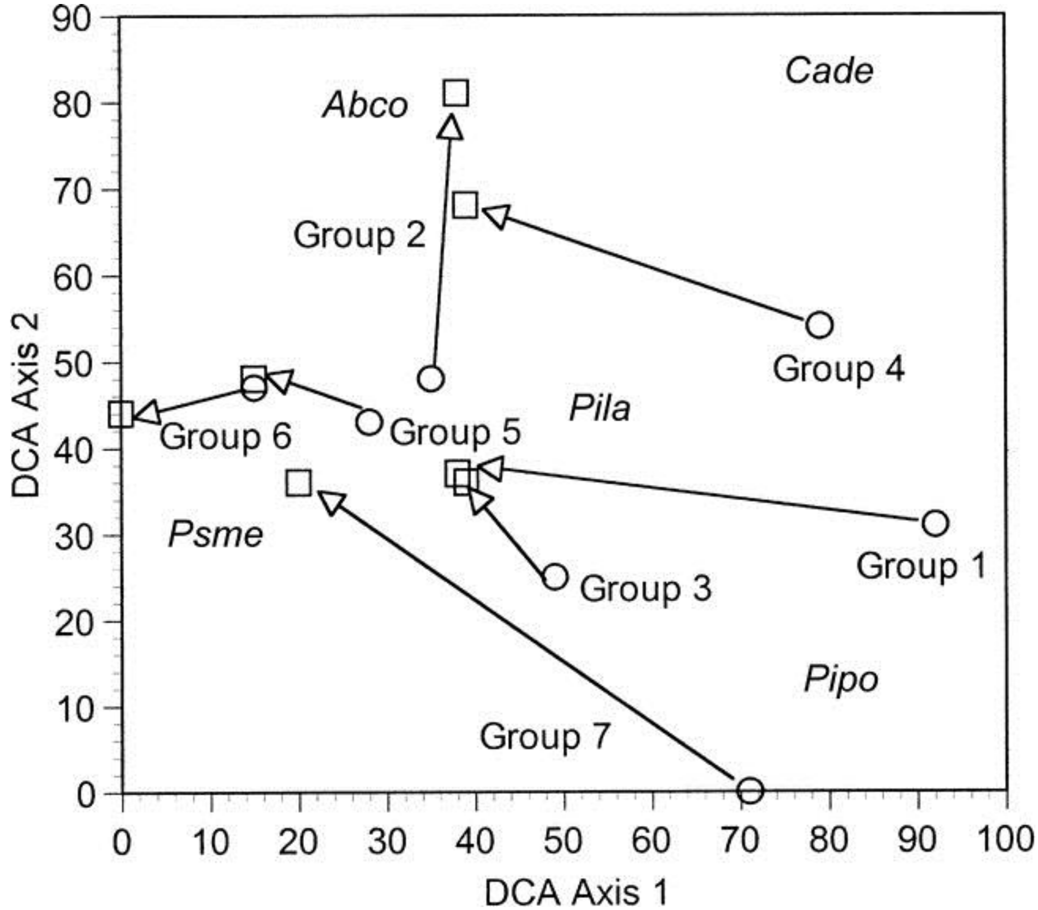

Figure 1.2. Compositional differences in the density (stems/ha) of <1l00-yr-old (squares) and >100-yr-old (circles) trees in the seven age-class groups identified by cluster analysis. Vectors show the direction and magnitude of compositional difference between the two age classes for each group in DCA species space. The position of species abbreviations represent regions of relative dominance. Species abbreviations are given in Table 1 (Figure and figure caption from Taylor and Skinner (2003)).

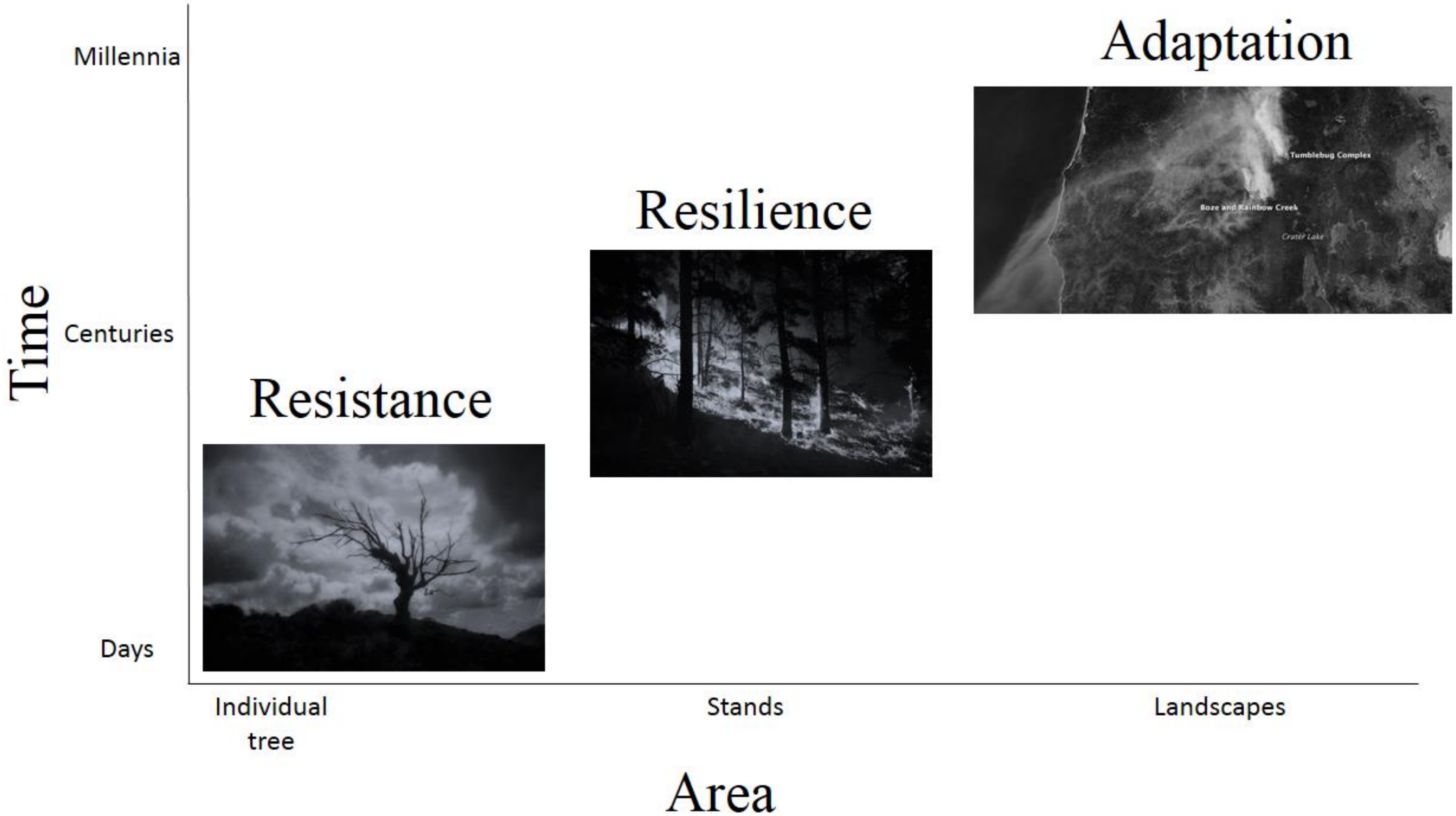

Figure 1.3. Conceptual model of multi-scaled fire regimes.

Table 1.1 Summary of fire frequency data from studies in the Klamath-Siskiyou and Southern Cascades region.

Citation	Study Size	Plot Size	Study Elevation	Group	Group Details	n plots	MFRI	Range low	Range high
Taylor and Skinner (2003)	2325 ha	150-2400 sqm (~20 trees/ plot)	840-1360m	*Elevation*	*range*				
				low	750-949 m	15	12.5	5.5	76
				middle	950-1149 m	45	14	7	60.5
				high	>1150 m	32	12	5.5	23
				aspect					
				North		35	16.5	5.5	76
				East		16	11.3	5.5	21
				South		11	12.5	7	22
				West		30	12	5.5	64
Taylor and Skinner (1998)	1570 ha	100-2250 sqm (~20 trees/ plot)	650-1600m	*Aspect*					
				North	315-44	7	15	8	54
				East	45-134	24	16.5	5	116
				South	135-224	11	8	4	35.5
				West	225-314	18	13	5.5	63
				Slope Position	*elevation range*				
				Lower	665-969 m	17	19	5	87
				Middle	970-1269 m	27	14	6.5	116
				Upper	1270-1569 m	16	10.5	4	37.5
Skinner (2003)		1-2 ha	1300-1750m	*plot*					
				RootCreek	Riparian	1	33	7	65
				RootCreek	upland	1	7	3	44
				N.F. Shotgun Cr	Riparian	1	16	5	56
				N.F. Shotgun Cr	upland	1	8	4	64
				Scott Camp Creek	Riparian	1	21	12	71
				Soapston Gulch	Riparian	1	42	9	52
				EFTA	Both	1	13	6	47
				EFTB	Both	1	13	4	47

Table 1.1 (con't)

Citation	Study Size	Plot Size	Study Elevation	Group	Group Details	n plots	MFRI	Range	
								low	high
Fry and Stephens (2003)	120 ha	1.4-1.7 ha	1150-1525m	plot					
				a		1	10.5	3	76
				b		1	10.5	5	76
				c		1	12.5	3	76
				e		1	13.5	5	76
				f		1	13	5	76
				g		1	10	3	76
Beaty and Taylor (2001)	1587 ha	0.04ha	1136 to 2044m	Aspect/Elevation			34	18	54
				north/ low		NR	17	14	26
				south/high		NR	13.5	5	43
				north/high		NR	9	43	25
				south/low		NR	7	1	21
Wills (1994)	~100 ha	5-8 ha	900-980m	site 1		1			
					pre-settlemt		19	5	23
					settlemt		9.5	6	20
					suppresion		27	16	52
				site 2		1			
					pre-settlemt		9.5	5	18
					settlemt		8	7	12
					suppresion		20.5	16	29
				site3		1			
					pre-settlemt		8.5	5	20
					settlemt		13	5	18
					suppresion		25	3	71
Comfort et al (in development)	34,000 ha	100 ha	340-1530m		prior to 1910	7	8.3	1	28
					1910 to current	7	46.9	11	>94

Table 1.2. Fire return interval for larger fires (generally >500 ha) listed by study.

Author	Fire Return Interval			
	Mean	Min	Max	Median
Taylor and Skinner (2003)	18.3	2	38	17
Taylor and Skinner (1998)	23.8	5	91	11
Beaty and Taylor (2001)	43.8	6	81	44
Briles et al. (2007)	105.0	60	150	105
Whitlock et al. (2004)	75.0	49	101	75
Whitlock et al. (2004)	32.2	6	94	24
Whitlock et al. (2004)	18.3	3	96	9.5
Whitlock et al. (2004)	23.7	4	53	24
Comfort et al. (In dev)	23.4	4	61	18

Table 1.3. Changes in the number of fire episodes estimated from charcoal peaks and background charcoal particles levels over time from lake sediment cores.

Author	Vegetation Association	Elevation (m)	Lake Size (ha)	cal yr before present	Fire Episodes/1000 years	Background Particles ($cm^{-2}yr^{-1}$)
Briles et al. (2005)	white fir	1638	5	14500 to present	4-10	
				14000	4	0.2-2
				13000	7	0.2-2
				11500	3	0.2-2
				10900 to 9200	8	3
				9200 to 8500	7	2.5-3
				8500 to 7000	10	4.5
				7000 to 3500	7	3.5
				3500 to 2500	8	5
				2500 to 1500	7	3.3
				1500 to present	9	
Briles et al. (2008)	white fir	1550	4	14500 to present	1.5-6.9	0.01-0.94
				14500 to 10400	1.5- 3.7	<0.2
				10400 to 9200	6.5	0.84
				9200 to 7200	4.4	0.84
				7200 to 5300	4.2	0.6- 0.8
				5300 to 2200	6.9	0.6- 0.8
				2200 to present	3.5	0.45
Mohr et al. (2000)	open pine	1921	NR	>13100	6- 9	0- 0.3
				13100 to 11100	5- 8	0-0.3
				11100 to 4450	6- 10	0.2- 1
				4450 ro 2150	5-8	0.1- 0.9
				2150 to present	6- 10	0.3- 1.5
	high elevation pine	2288	NR	8400 to 5650	11-17	0- 0.2
				5650 to 2150	10-13	0.1- 0.9
				2150 to present	9- 13	0.03 to 0.9
Colombaroli and Gavin (2010)	mixed evergreen	980	7.2	2000 yrs to present		
	mesic mixed-conifer			AD 200-550	fire-free	NR
				AD900	largest pre-settlement peak	NR
				AD950- 1450	5 peaks	NR
				1450 to 1900	fire infrequent	NR
				1900 to present	high fire	NR

CONTRIBUTION OF AUTHORS

This research was designed and written by Emily Comfort under the guidance of John Bailey.

CHAPTER 2: FUEL, FOREST STRUCTURE, AND FIRE SEVERITY HETEROGENEITY ACROSS EDGES AND OWNERSHIP BOUNDARIES IN THE ELK CREEK WATERSHED, OR

Abstract

Fires move across landscapes in response to fuels, topography, and weather conditions. Fire creates patterns on the landscape through its severity and it responds to patterns created by past disturbance and management. In dry, mixed-conifer forests in southwestern Oregon the historic mixed-severity fire regimes typically fostered a patchy distribution of fire severities and resulting age-class variety at multiple scales. The current landscape has a complex ownership pattern that interacts with disturbance process and creates highly variable edge structures. There are both abrupt edges, where there is a large difference in age between canopy cohorts and a distinct change in forest structure across stand boundaries and diffuse edges where there are smaller differences in both age and structure. I measured fuel structure at 12 abrupt edges and 13 diffuse edges in the Elk Creek watershed in southern Oregon to examine local-scale pre-fire differences in fuel structure. I also analyzed landscape-scale fire effects across ownership boundaries and age-class edges using remotely-sensed data to examine landscape-scale implications of different edge structures. I found that there is very little difference in fuel structure at the stand-level between abrupt and diffuse boundaries. At the landscape-scale, however, a patchy distribution of dominant-tree age classes tended to be associated with lower-severity fire than more uniform dominant-tree age distributions. Historic fire regimes in this area created a self-reinforcing patchy distribution of age classes, with small patch sizes of high-severity fire within a matrix of low- and moderate- severity fire. Timber management, including salvage-logging, may be increasing the patch size of high-severity disturbance through homogenization of structure and the edge effect on the surrounding forests.

Introduction

The interaction between pattern and process is a central theme in landscape ecology (Turner 1989). In mixed-severity fire regimes common in southwestern Oregon, the patterns of fire severity are considered self-reinforcing (Odion et al. 2004, Thompson et al. 2007, Perry et al. 2011). Edge effects and edge influence are often used to describe patterns in biodiversity (Baker et al. 2013) or compositional changes within patches (Chen et al. 1992, Harper et al. 2005), but not as a central driver of fire severity patterns. Different edge structures, however, have different influences on many ecological processes (Knight and Landres 1998), including fuel loads and other factors that influence spread and intensity of wildfire as well as post-fire community composition. This research examines the influence of edge structure, defined by gradients in age structure and ownership boundaries, on surface fuels loads and fire behavior in order to explore the drivers of fire severity in mixed-ownership landscapes.

Forest landscapes in southwestern Oregon are characterized by a complex ownership pattern. Publicly owned land is distributed across the landscape both as small patches that are interspersed with small patches of privately owned timberland and as larger, contiguous patches. Federal land managers have to balance multiple land-use goals including conservation of protected species' habitat, fire risk reduction in the wildland urban interface, and timber production. Private industry land is generally intensively managed for timber products. This pattern of ownership has implications for fire behavior and fire effects. Prior research is equivocal about the role of management activities in mediating fire effects (Omi and Kalabokidis 1991, Weatherspoon and Skinner 1995, Thompson et al. 2007). The direction and magnitude of fire contagion from public lands to private industry lands and vice versa are not well understood. The structure of fuels at patch edges may increase our understanding of how fire spreads across mixed-ownership landscapes.

Prior to Euro-American settlement, both bottom-up and top-down controls on fire would have created patches that differed in age and composition based on time since the last fire and fire severity (Hessburg et al. 2000, Hessburg and Agee 2003, Hessburg et al. 2005). These edges most likely would have followed physiographic features on the landscape, rather than arbitrary property boundaries (Heyerdahl et al. 2001, Gavin et al. 2006) and had self-reinforcing properties (Holling et al. 1996). Following 150 years of forest change due to changes in land use practices and shifting fire regimes (Taylor and Skinner 1998, Beaty and Taylor 2001, Skinner 2003, Taylor and Skinner 2003, Borman 2005), dry forests in southwestern Oregon are thought to have greater continuity of forest structure and fuel structure (Skinner 1995, Hessburg et al. 2005, Comfort et al. *In Preparation*). Additionally, climate change may result in further changes including longer fire seasons (Westerling et al. 2006) and a shift from top-down (climate) control to bottom-up (fuel) control (Littell et al. 2009). However, the resulting changes in fire effects will be context dependent (both management and past fire history) (Stephens et al. 2013).

Edges are often measured as distinct features on the landscape (Chen et al. 1995, Hargis et al. 1998). In most undeveloped landscapes, however, edges are not easily defined. There are hard edges that are created by disturbance events such as high-severity fire or logging where the disturbance is adjacent to mature forest (Figure 3.1). More diffuse edges are also common where less severe disturbance occurs or as hard edges age (Figure 3.1). These different kinds of edges create fundamentally different fuel loads and forest structures that potentially impact fire behavior. Approaches that incorporate the gradient nature of these categorical variables have not been well developed (McGarigal and Cushman 2005, Cushman et al. 2010). Additionally, it has been difficult to incorporate metrics that separate the effects of patch edge from patch interior (Betts et al. 2006).

The purpose of this research was to determine if spatial patterns of fire severity were reinforced by edge effects from management/disturbance and ownership. I used two approaches. At the stand level, I wanted to determine if structurally-dissimilar edge types influenced community composition and surface fuel structure. Also, I wanted to determine how far into a patch each edge type extended its influence. I used field data from a mixed-ownership landscape that had recently experienced a mixed-severity disturbance to test for these differences in edge types. I also wanted to determine if edges impacted fire behavior directly. I modeled the relationship between disturbance severity following an 11,000 ha fire and subsequent salvage logging and a calculated measure of landscape age-structure that matched our definition of edge from the field. Finally, I wanted to determine if ownership boundaries operated in the same way as structural edges, so I modeled disturbance severity as a function of distance from ownership boundaries.

Methods

Study Area

The study area is the Elk Creek 5th field watershed in southern Oregon (Figure 3.2.). It is primarily managed by the Forest Service, BLM, and private forest industry. There is a range of management objectives and activities from late-successional reserves on public lands to even-aged plantations on private industrial forest land. The study area is approximately 34,500 ha and steep, dissected, complex terrain. Elevations range from 445 m to 1765m. Precipitation ranges from 127 to 381 cm yr^{-1} and occurs mostly during the winter and spring. Temperatures range from 0° C to 40° C. The study area includes two ecoregions, the Klamath Mountains and Southern Cascades (McNab and Avers 1994), and hence has a high level of ecological diversity. Dominant vegetation associations include interior valley (*Pinus- Quercus-Pseudotsuga*), mixed-evergreen (*Pseudotsuga*-Sclerophyll), mixed-conifer (*Pinus-Pseudotsuga-Libocedrus-Abies*), *Abies concolor,* and *Abies magnifica shastensis*. Historical disturbances include fire, flooding, pests, disease, and windthrow. In 2002, the Timbered Rock fire burned 11,000 ha within the Elk Creek watershed.

Composition and surface fuel structure at edges

In the summer of 2010, I collected fuel and forest structure data at 30 paired-plots at abrupt and diffuse edges (Figure 3.3). Edges within the watershed were classified as abrupt or diffuse using an agency-provided map of stand ages. Abrupt edges were defined as the interface between stands less than 20 years old and stands greater than 40-year age older. Diffuse edges were defined as the interface between stands that had less than a 40-year age difference. Representation across the watershed was accomplished by 1) randomly selecting paired plot locations from all available stand boundaries in the Elk Creek watershed that were safely accessible (less than 35% slope) and 2) systematically implementing the field work across the watershed. I randomly selected 100 edges (50 abrupt and 50 diffuse) using Hawth's Analysis Tools for ArcGIS (Beyer 2004) and arbitrarily assigned somewhere along each edge where slope was < 35% to serve as the paired-plot center in ArcGIS. Paired plots were grouped into overlapping clusters of 3 to 5 adjacent paired plot centers prior to entering the field. For each cluster, the field crew visited paired plots in the order provided by the GIS ID assignment. If a paired plot was disqualified because 1) it was too steep to safely sample or 2) the edge was misclassified in the GIS layer (i.e. more recent management/ disturbance than identified in the GIS or management that was listed had not occurred), we moved in the designated order to the next paired plot in that cluster. I sampled the first paired plot in each cluster that met criteria and did not sample any other paired-plots in that cluster. Of the 30 plots we sampled, one plot was later rejected because it was incorrectly located in the field (not on an edge) and three plots have incomplete fuels data because the data recording device broke in the field. These plots were not used in this analysis. At each paired plot, the older side of the edge was labeled the mature side and the younger side was labeled the disturbed side.

Forest Structure Measurements

We conducted standard tree inventories on each side of the paired-plot in order to assess structural changes across edges (Figure 3.3). We recorded species, diameter at breast height (cm), height (m), and height to live crown (m) in a 20 m X 50 m (0.1 ha) plot on each side of the edge. We tallied the number of small trees (<1.4 m in height) by height class and species in 5m X 20m (0.01 ha) plots in the center of the 0.1 ha plots.

We calculated total basal area, average tree height and average live crown length of trees (>1.4 m in height) and basal area by species for both the mature side and the disturbed side of each paired plot for the 0.1 ha plots. We calculated average density of trees less than 1.4 m in height in the 0.01 ha plots.

Forest Structure Statistical Analysis

To verify that forest structure also varied by my age-based definition of edge type, as the study design intended, I tested whether the absolute difference in each structure variable from the mature side to the disturbed side varied by edge type. Paired plots were also expected to vary, based on different ecological (forest type) and physiographic conditions (elevation, aspect, heat load). I tested for a difference in the mean value of my structure variables between the two sides of the paired-plot because positive covariance will increase the precision of the estimate of the difference between two means. I expected the largest differences to be at abrupt edges.

I used a Multi-Response Permutation Procedure (MRPP, McCune and Grace, 2002) to test the relationship between edge type and community composition using Sorensen's distance measure. The species matrix (58 plots, 20 columns: 19 tree species, snags) contained the total basal area of each tree species present in the 0.1 ha plot. I did not identify any species as an outlier. However, I did identify one plot as an outlier. It was a plot with very low basal area on the disturbed side of the paired plot, but I considered it valid data, so I retained it in the analysis. The MRPP analysis tested the following four edge-subplot group assignments: abrupt edge, mature plot (AM); abrupt edge, disturbed plot (AD); diffuse edge, mature plot (DM); diffuse edge, disturbed plot (DD). I expected community composition to vary with group and the largest differences to be between the AM and AD group and the smallest difference between the DM and DD group. I used an indicator species analysis to determine which species were closely associated with each group. All multivariate analyses were run in PC-ORD version 5.18 (McCune and Mefford 2006).

Fuel sampling

At each paired plot, I measured surface fuels on 30 m transects located parallel to the edge at 5 m, 10 m, 25 m and 50 m from the edge (Figure 3.4). Using methods adapted from Brown (1974), we measured fine woody fuel, live and dead herbaceous and shrub fuel, and litter and duff fuel data at each transect and measured slope (%). On a 6 m section at one end (arbitrarily assigned prior to entering the field) of each 30 m transect, we counted all dead woody material less than 0.6 cm in diameter (1-hr fuels) that intersected the transect using a go-no-go gauge to determine diameter. On the following 12 m section of each transect, we counted all dead woody material 0.6 to 2.5 cm in diameter (10-hr fuels) than intersected the transect. On the following 12 m section of each transect, we counted all dead woody material 2.5cm to 7.6 cm in diameter (100-hr fuels). I recorded herbaceous cover and height and litter depth at two 0.5 m^2 herbaceous fuel subplots located at 20 m and 40 m along each transect. I estimated shrub cover and height at two 3.1m^2 shrub subplots located 15 m and 30m along each transect. Finally, we measured duff depth at 1 m and 6 m along each transect.

I calculated fuel loads (kg m^{-2}) using methods developed by Brown (1974), but adapted to my sampling design. From the field data, I used the slope of the fuels transect and the number of dead woody stems in each size class. Additional information that I needed to calculate fuel loads were obtained from the literature: 1) average diameter of stems in each size class, 2) specific gravity for each size class for conifers and 3) adjustments for non-horizontal position of some woody material. Fine woody fuels are the aggregated volume of 1-hr, 10-hr, and 100. I estimated herbaceous fuel loads and shrub fuel loads for each transect from the average of the two percent covers and heights recorded at each transect in the field and bulk density estimates obtained from the literature. Live fuels are the aggregated volume of herbaceous fuels and shrubs fuels. Finally, I estimated the volume of litter and duff fuel loads for each transect using the average of the two depths collected in the field and bulk density estimates from the literature.

Fuels statistical analysis

Fuels were expected to vary by edge type (abrupt or diffuse) and distance from the edge. Paired plots were also expected to vary, based on different ecological (forest type) and physiographic conditions (elevation, aspect, heat load), and were included in analyses as a random effect. Differences in fuel loads across the edge types were expected to be correlated within paired plots with closer measurements being more correlated than more distant measurements. Random variation will affect results, but because paired plot locations are dispersed through the study area at randomly selected sites, I expected the random variation to be unbiased. Twenty-five paired plots were used in this analysis, 13 were on diffuse boundaries and 12 were on abrupt boundaries. Four fuels transects were located in each half of the paired plot, so there are four measurements of the difference in fuel load for each paired plots. In total, I had 52 observations (13 paired plots with 4 repeated measures) at diffuse edge and 48 (12 paired plots with 4 repeated measures) at abrupt edges.

A linear mixed effects model with a compound symmetry correlation structure was used to compare boundary types and distance from boundary. Several correlation structures were investigated, but the compound symmetry fit the data best as assessed using the Baysian Information Criterion (BIC) value. BIC rates models based on fit to data, number of parameters, and sample size (Johnson and Omland 2004, Ward 2008).

Specifically, I tested the difference in fine woody fuel loads, herbaceous fuel loads, and litter and duff fuel loads across edges and determined 1) if the difference in fuel load across the edge (mature side to disturbed side) averaged over all 4 transect distances was different between edge types (main effect of edge type), 2) if the relationship between the difference in fuel across the edge and distance from edge (5, 10, 25, and 50 m) varied by edge type (main effect of distance and edge type), and 3) if there was a critical transect distance from the stand edge at which the difference in fuel load across edges changes in one edge type by not the other (interaction between edge type and distance). I considered a difference of 0.2 kg m-^{2} to be biologically significant. Preliminary analyses using custom fuel models suggested that a change in 1-hr fuel load of 0.2 kg m^{-2} almost doubled the rate of spread and increased the flame length of potential fire under moderate fire weather conditions, which would contribute to a major change in fire behavior across stand boundaries. I expected this relationship between fuel load and fire behavior to hold for the surface fuels examined in this study.

Edges and fire effects

I used GIS data from a recent fire in the study area. The 2002 Timbered Rock Fire burned 11,000 ha on both federally managed land and private industry land within the Elk Creek watershed. In ArcGIS (version 10), I generated 10,000 random points within the fire boundary of the Timbered Rock Fire. I used fire severity maps from Monitoring Trends in Burn Severity (MTBS) (Eidenshink et al. 2007) that depict disturbance severity. Normalized burn index uses near infrared band 4 and short wave infrared band 7 from Landsat data to estimate the vegetation condition across the landscape. Disturbance severity is the relative difference in normalized burn index from before and after the Timbered Rock fire. Because salvage logging was conducted on private industry lands immediately following the fire, “fire severity” in this analysis is indistinguishable from salvage-logging severity. I also used normalized burn index from the pre-fire image to estimate pre-fire vegetation condition. In the field, I used stand age to assign stand boundaries into two categories, abrupt and diffuse. In order to estimate a similar metric at the landscape scale, I used Gradient Nearest Neighbor (GNN) (Ohmann and Gregory 2002) data from before the fire (2000). I created a raster layer with the age of dominant trees prior to the Timbered Rock Fire. I then calculated the slope of change in age of dominant trees across space using the slope calculator in ArcGIS 10 (Appendix 3). This procedure created a raster layer with the slope of change across space (% change) in age at each 30m by 30m pixel on the map. The procedure is similar to creating a slope of elevation map. Small values are indicative of flat age structure (even-aged) and high values are indicative of large differences in age over the same spatial scale (un-even aged).

I used a general linear model to test for the relationship between disturbance severity and the slope of change in age structure. I expected the relationship to vary by ownership (public or private), pre-fire condition, and canopy age, so I included these as covariates. Pre-fire condition was log-transformed and disturbance severity was square-root transformed in both models to meet model assumptions. Neither age nor any interactions with age were significant and it was not a variable of interest, so it was dropped from the final model.

I also wanted to compare disturbance severity at ownership boundaries. I used a map of ownership boundaries to estimate the distance from ownership boundaries for each random point. Because the largest contiguous extent of publically owned lands was greater than that of privately owned land, I eliminated random points that were located on federal lands farther than the farthest distance for private lands (1000m). I also eliminated random points that increased in greenness from before to after the fire (negative values of RdNBR). The final number of random locations was 7144.

I used a general linear model to test for the relationship between disturbance severity and the slope of change in age structure. I expected the relationship to vary by ownership (public or private), pre-fire condition. Distance was log-transformed, pre-fire condition was log-transformed, and disturbance severity was square-root transformed in both models to meet model assumptions.

Results

Composition and surface fuel structure at edges

Overstory structure

The difference in structure at abrupt and diffuse edges confirmed that I were able to capture two fundamentally different edge-types (Table 2.1). There was strong evidence that the difference in basal area from the mature side to the disturbed side of a paired plot was different between abrupt and diffuse edges ($F_{1,27}$ =8.39, p= 0.0074). On average, basal area was 3.9 m^2 (95% CI: 0.8 to 4.5 m^2) greater on the mature side than on the disturbed side of abrupt edges and 1.2 m^2 (95% CI: -0.1 to 2.6 m^2) greater at diffuse edges. The number of trees in each plot did not vary by edge type ($F_{1,27}$=0.0.19, p=0.8915) in my study. There was moderate evidence to suggest that the difference in height from the mature side to the disturbed side of a paired plot varied between edge types ($F_{1,27}$=6.35, p= 0.0179). On average, tree height was 5.6 m (95% CI: 0.9 to 8.5 m) taller on the mature side than on the disturbed side of abrupt edges and 0.9 m (95% CI: -1.8 to 3.6 m) taller at diffuse edges. There was strong evidence to suggest that there is a difference between edge types in the difference between maximum tree height from the mature side to the disturbed side ($F_{1,27}$=16.1, p< 0.001). On average, maximum tree height was 30.6 m (95% CI: 10.7 to 33.1 m) taller on the mature side than on the disturbed side of abrupt edges and 8.6 m (95% CI: 0.6 to 16.7 m) taller at diffuse edges. There was moderate evidence to suggest a difference between abrupt and diffuse edges in the mean difference in live crown ratio ($F_{1,27}$=4.7, p= 0.0391). On average, live crown ratio was 0.19 (95% CI: -0.19 to -0.01) smaller on the mature side than on the disturbed side of abrupt edges and 0.09 (95% CI: -0.15 to -0.02 m) smaller at diffuse edges.

Overstory composition

The edge-subplot group assignments - abrupt edge, mature plot (AM); abrupt edge, disturbed plot (AD); diffuse edge, mature plot (DM); and diffuse edge, disturbed plot (DD) - predicted community composition from the species basal area well

(MRPP: A=0.09, p= 0.0002) (Table 2.2). The pairwise comparison of groups suggested that strongest difference in species community was between the AO group and the AR group (A= 0.17, p <0.0001). There was a slight difference in composition between both mature groups, AR and DR. There was no evidence of a difference in species composition across diffuse boundaries (A= 0.03, p=0.0590). Pacific yew was only found at diffuse edge plots, but was rare throughout the study, so it was not a significant indicator of diffuse edge groups. Douglas-fir was common throughout the study area, but was a statistically significant indicator for the AR group (p= 0.0016).

There was no evidence in this study for a difference between the mean number of small trees (<1.4 m in height) across stand boundaries between abrupt and diffuse stand boundaries ($F_{1,26}$= 1.22, p = 0.2787).

Fine woody fuels

I found no detectable difference in fine woody fuel volume between edge types ($F_{1,23}$=0.18, p = 0.8931) or among transect distances from the edge ($F_{3,69}$= 1.03, p = 0.3866). There also was no evidence of an interaction between the edge type and the transect distance from the edge ($F_{3,69}$=0.56, p = 0.6442) (Figure 3.5). The estimated difference between fine woody fuel loads included both negative and positive biologically significant values at each transect distance and average over all transects. Difference values are reported in the Appendix (Tables A3.1, A3.2, and A3.3). The mixed-effects model to describe the difference in fine woody fuel loads met model assumptions of constant variance and normality. There was one outlier, but removing it did not affect results, so it was retained in the analysis.

Live fuels

I also found no detectable difference in live fuel loads between the mature side and disturbed side of abrupt versus diffuse edges ($F_{1,23}$=0.74 p = 0.3971) or among transect distances from the edge ($F_{3,69}$= 0.23, p = 0.8734). There also was no evidence in this study of an interaction between edge type and the transect distance from the edge ($F_{3,69}$=0.48, p = 0.7001) (Figure 3.6). The estimated mean difference suggests there are more live fuel in general on the disturbed side than on the mature side, however none of the average differences are biologically significant values and the 95% confidence interval for the true mean overlaps 0. Difference values are reported in the Appendix (Tables A3.4, A3.5, and A3.6). The mixed effects model to describe the difference in live fuel loads met model assumptions of constant variance and normality. There were two outliers, but removing them did not change the coefficients for the model (it slightly increased the F-value for the effect of transect distance class) and there was no discernible recording error or biological reason to remove them, so it was retained in the analysis.

Litter and duff fuels

Finally, I found no detectable difference in litter and diff fuel load between abrupt and diffuse edges ($F_{1,23}$=2.08 p = 0.1624) or among transect distances from the edge ($F_{3,67}$= 2.29, p = 0.0864). There also was no evidence in this study of an interaction between the edge type and the transect distance from the edge ($F_{3,69}$=0.14, p = 0.9367) (Figure 3.7). The 95% confidence intervals for all mean differences overlap 0 and many have both positive and negative biologically significant values. Differences and figures are reported in the Appendix (Tables A3.7, A3.8, and A3.9). The mixed effects model to describe the difference in litter and duff fuel loads met model assumptions of constant variance and normality. There was an outlier and removing it changed coefficient for the effect of the transect at 25 m from the edge and doubled the F-value for the effect of edge type; however, the outlier was a biologically plausible value, so it was retained in the analysis.

Edges and fire effects

Although I found little evidence for differences in fuel structure at diffuse and abrupt edges, fire effects were strongly related to the rate of change in age structure over space (Figure 3.8). The model to test the association of disturbance severity and rate of change in age structure with the best fit (BIC=46771 with no competing models) included significant interaction effects between ownership and rate of change in age structure ($F_{1,7138}$=7.7882, p=0.0052) and ownership and pre-fire condition ($F_{1,7138}$=29.2584, p<0.001). There was no interaction between rate of change in age structure and pre-fire condition. The results suggest that disturbance severity was related to both pre-fire condition and rate of change in age structure, but that the relationship was different on public and private land.

There was a decrease in disturbance severity with increasing rate of change in age-structure over space within public lands. On average, disturbance severity decreased 129 units (95% CI: 105 to 153 units) as the rate of change in age-structure over space increased from 0 to 600, or as the patchiness of age-structure in a 0.8 ha neighborhood increased up to 6 times. This decrease represents 12% of the range of fire severity at the random points. Prior to the fire, ages were fairly constant and independent of distance from stand boundary on private land. Ages were more variable and slightly greater on publicly owned land (Figure 3.9).

The model to test the association of disturbance severity and distance from ownership boundary with the best fit (BIC= 17431, with no competing models) had a significant interaction effect for ownership by distance ($F_{1,7139}$=43.4967, p <0.001) and for ownership by pre-fire condition ($F_{1,7138}$= 9.7956, p=0.0018). There was no interaction between distance and pre-fire condition. The results suggests that disturbance severity was related to both pre-fire condition and distance from ownership boundary, but that relationship was different on public land and private land.

On private land, salvage logging increased the overall disturbance severity and decreased the range of values. Because salvage logging is confounded with fire severity, there does not appear to be a relationship between disturbance severity and either distance to ownership boundary or rate of change in age structure, but there is higher disturbance severity across the range of both distance from ownership boundary and rate of change in age structure than on public lands. On public lands, no salvage-logging occurred and there is a strong relationship between disturbance severity and both distance to ownership boundary and rate of change in age structure. On public land, disturbance severity decreased with distance from ownership boundary with the largest decrease in the first 250 m (decreased 104 units, 95% CI: 87 to 124 units) followed by continued slow decline (decreased another 50 units (95% CI: 38 to 54 units). The range of fire severity across all the random points used in this analysis was 1115, so the observed average decline in the first 250 m represents 9% of the range of disturbance severity (Figure 3.10).

Discussion

Stand-level changes in forest structure

I confirmed that my definition of diffuse and abrupt edges captured very different gradients in stand structure and community composition across the edge boundary, as the study design intended. The differences in basal area, height, and live crown ratio confirm that my definitions of edge (abrupt edges have greater than 40 year difference in age and a young stand on the disturbed side, diffuse edges have less than 40 year difference in age) captured two distinct changes in forest structure. Additionally, community structure of basal area by species grouped well by edge type. This result suggests that in addition to having unique physical structure, there are also different local environment and competition drivers at abrupt and diffuse edges. While these gradients have been studied at abrupt edges (Chen et al. 1995, Davies-Colley et al. 2000). I am not aware of similar studies for diffuse boundaries; however, Comfort et al. (2010) found that established midcanopy trees in second growth forests responded to variable-retention harvests, which may approximate my diffuse edge type. Midcanopy trees increased growth following harvest in both thinned and small retention patches suggesting that there is an edge effect on resource availability from a thinned stand into small retained patches. In this study, there was not a difference in density of small trees on either side of diffuse and abrupt edges. At abrupt edges, the increase in resource availability may be promoting the natural regeneration of small trees on the older mature side (Chen et al. 1992, Harper and Macdonald 2002) that approximately equals the amount of regeneration through planting and/or natural regeneration on the younger disturbed side. At diffuse stand boundaries, both sides are generally either in a fairly open canopy stage or closed canopy stage and hence the same amount of available growing space for natural regeneration.

As expected, the most distinct differences in community structure were between paired 0.1 ha plots at abrupt edges (Harper et al. 2005). There was also greater homogeneity within groups than between the mature-side 0.1 ha plots for both abrupt and diffuse edges. This could suggest that drivers of community structure operate differently at diffuse and abrupt edges. Light is more available on the mature side of abrupt stand boundaries, which may increase the importance value of good competitors like Douglas-fir. The more subtle changes in competitive environment at diffuse stand boundaries may benefit more shade tolerant species like Pacific yew. Alternatively, the difference in community structure in mature-side edge plots may suggest that abrupt and diffuse edges are not randomly placed on the landscape, but are more likely under certain physiographic conditions that also coincide with distinct species communities. There was little difference in community structure across diffuse stand boundaries. This may indicate that edge effects are minimal. Vegetation communities were significantly closer to old-growth community structure in edges between unsalvage-logged edges and salvage-logged edges 14 years following a wildlife in a northern California mixed conifer-hardwood stand (Hanson and Stuart 2005). While we did not look at fire or salvage logging created edges in particular in this study, the two edge types would fall into our categories of diffuse and hard edge respectively.

Stand-level changes in fuel structure

This study provided no evidence that fine woody fuel loads, live fuel loads, or litter and duff fuel loads are different for either abrupt or diffuse edges in these types of complex forested landscapes. The wide range of possible mean values we found suggests that fuel load estimates are not precise enough to detect either a difference or the direction of difference. Additional replication might have increased the precision of the estimate a mean fuel differences, but fuel loads were highly variable in this and similar landscapes at spatial scales as small as 1-5 m (Keane et al. 2012, Keane et al. 2013) because fine fuels are highly correlated with vegetation structure (Keane 2008). Controlling variability with sampling design would likely limit the scope of inference.

For example, sites could be chosen that represent a more limited set of biophysical characteristics, such as one forest association or a more narrow range of slope, aspect, and elevation. This solution would further reduce the scope of inference of the study, but may provide more meaningful results for that limited set of biophysical conditions (Morgan et al. 2001, Rollins et al. 2004).

Finally, another explanation for the lack of relationship is that the difference is not apparent at the spatial scale used in the study (i.e. within 50m of stand boundaries) despite a literature suggesting that smaller scales are appropriate (e.g., Hanson and Stuart (2005). All mean values were below -0.20 (the biologically significant value). While the confidence intervals include both non-statistically and biologically significant values, this trend might suggest that at distances greater than 50 m abrupt stand boundaries have higher fuel loads on the disturbed side than diffuse stand boundaries. Interior stand conditions on either side of the boundary may or may not have different fuel loads depending on stand histories (e.g. site preparation and stand density). That is, at transect distances classes less than 50 m, fine woody fuels, live fuels, and litter and duff fuels are influenced by edge effects as well as similarities or differences between interior conditions. At the transect distance class of 50 m, the edge effects diminish and any difference in fuel loads between different stand types emerge. Donato et al. (2013) found that surface fuel mass increased with increasing post-fire logging intensity in the 3-4 years following fire. Patch interiors are likely to have different fuel loads that are self-reinforcing following disturbances (Odion et al.

2004, Stephens and Moghaddas 2005, Thompson et al. 2007) leading to similar fire severity patterns in future fires.

Edges and fire effects

In the Timbered Rock fire, disturbance severity and size are not consistent with historical fire regimes. Prior to Euro-American settlement, fire sizes were generally smaller than the 11,000 ha Timbered Rock Fire (Beaty and Taylor (2001), Taylor and Skinner (2003) reported median fire sizes of 106 and 128 ha respectively, with less frequent fires over 500 ha, for nearby study areas). The smaller fires would likely have had patches of high-severity fire at very small extents (<0.2 ha (Scholl and Taylor 2010) for a study area in Yosemite National Park). The average size of a privately owned "patch" in the Elk Creek watershed is 541 ha. We found that disturbance severity was higher on private land than on publically owned land. Under a high-severity fire regime, Omi and Kalabokidis (1991) found that fire severity was generally higher in unmanaged than in managed stands, but that landscape context also influenced severity. Because salvage logging occurred on most of the private industry land, our disturbance severity measure indicates artificially high and uniform severity. However, because of intensive management, private industry land is also likely to recover closed canopy forest more rapidly than federally-managed lands that experienced high-severity fire.

The relationship between fire severity and ownership suggests that there is a moderately significant increase in fire severity and/ or disturbance impacts from salvage logging within 250m of ownership boundaries. Similarly, Brudvig et al. (2012) found an increase in fire severity along corridor edges. They suggested that increased litter and solar radiation at edges could be responsible. While our GIS study did not include a measure of structure along the edge, the post-fire pattern suggested that either 1) the difference in forest and/or fuel structure across ownership boundaries was correlated with increased fire severity along ownership boundaries, or 2) there were impacts of salvage logging that extend into the adjacent unsalvaged forest. In two moist forest on the west slope of the Cascades, interior forest edges adjacent to clearcuts were characterized by decreased canopy cover, tree density, and basal area and an increase in mortality (number of fallen dead trees) compared to the forest interior within 10-15 years (Chen et al. 1992); edge effects were generally strongest within 12 m of edges. Hanson and Stuart (2005) found an increase in edge influence of 15-30 m in edges created by adjacent low- and high-severity fire when the high-severity portion had been salvage-logged compared to unlogged edges. Our results suggest that edge effects from salvage-logging, increased the patch size of higher severity disturbance.

Conclusions

Ecological forestry practices are being developed that attempt to emulate the pre-Euro-American settlement patchiness of dry forest (Franklin and Johnson 2012) and emphasize landscape-scale management plans that incorporate heterogeneity in the timing and intensity of management practices. However, the assumptions behind historical ranges of variation that would determine patch sizes are debated (Williams and Baker 2012), and the configuration of public and private land may impede implementing a landscape-scale management plan that includes fire. Our results suggest management practices that focus on maintaining a matrix of diverse age-classes may be a good strategy for reducing high-severity fire risk. This study does not show a difference in surface fuel across two different structures of edges at small scales (up to 50m), suggesting that surface fuels alone are not responsible for observed increases in fire severity. Fuel reduction treatments that treat only surface fuels and do not consider the larger landscape context therefore may not be effective at reducing the spread of higher-intensity fire across stand and ownership boundaries.

Private lands that are interspersed with public lands provide valuable services, such as sustainable timber supplies that reduce the pressure to harvest timber on federal lands. Their impact on landscape-scale processes reach beyond their boundaries, however, to affect landscape properties like patch size and edge structure.

On federally-managed lands that were not salvage logged, fire severity was greater close to the edges of boundaries with privately managed land. The difference in severity was most likely due to edge effects from salvage logging and not from the pre-existing fuel loads or the fire *per se*. However, this may increase the patch size of future high-severity disturbance on the landscape, which is already likely larger than historic patch size. This should be considered when devising landscape management plans. Small-scale patchiness in age-classes was associated with lower- and mixed-severity fire in pre-settlement forested landscapes, and are prominent still on federally-managed lands. So, fire exclusion has increased within patch homogeneity; large harvesting blocks have increased patch size – both can be a problem in terms of long-term resilience. While more research is needed to determine the drivers of reduced severity, our findings support management that restores a pattern of mixed age-classes.

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Figures

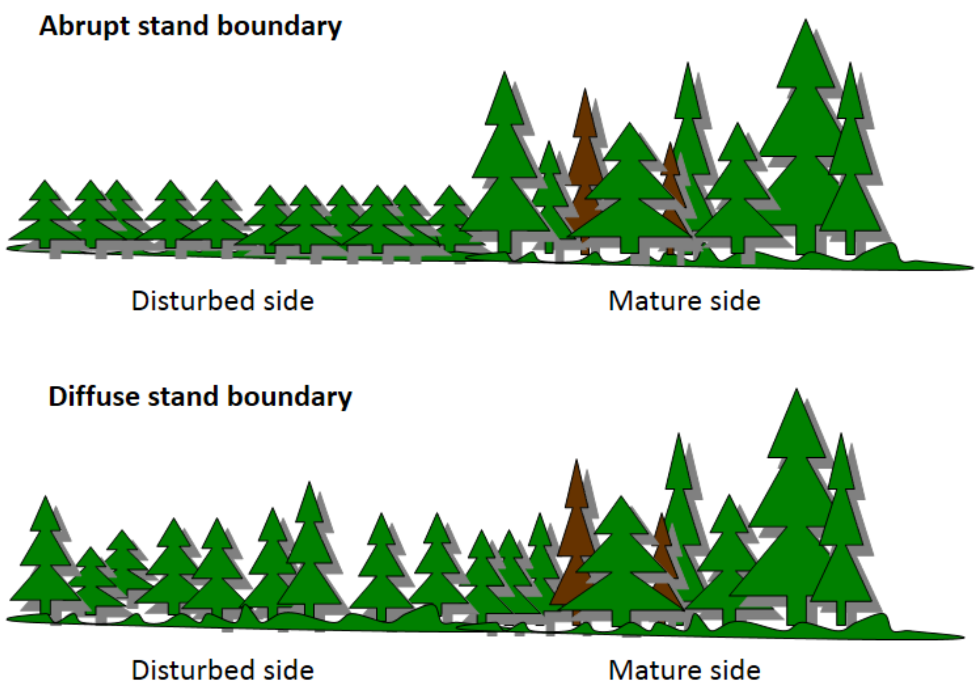

Figure 2.1. Conceptual illustration of abrupt versus diffuse stand edges.

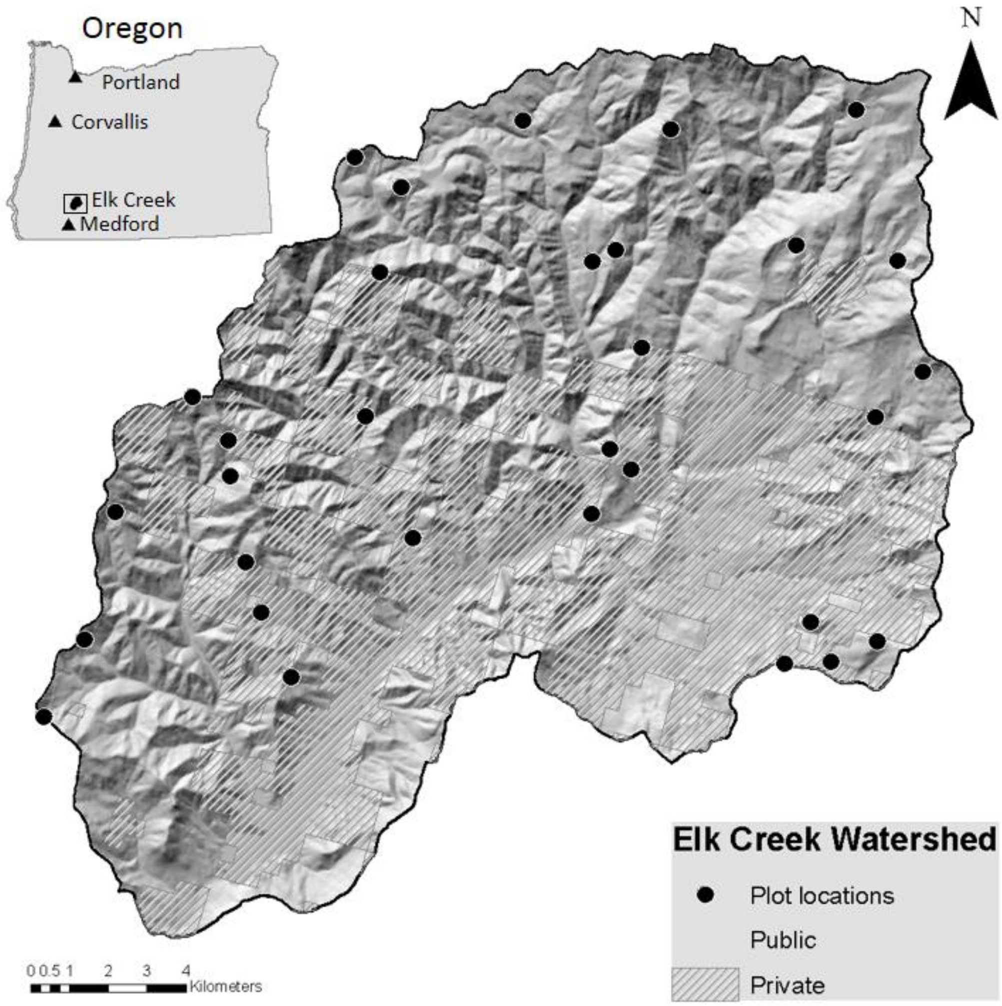

Figure 2.2 Map of study area with paired-plot locations and ownership boundaries.

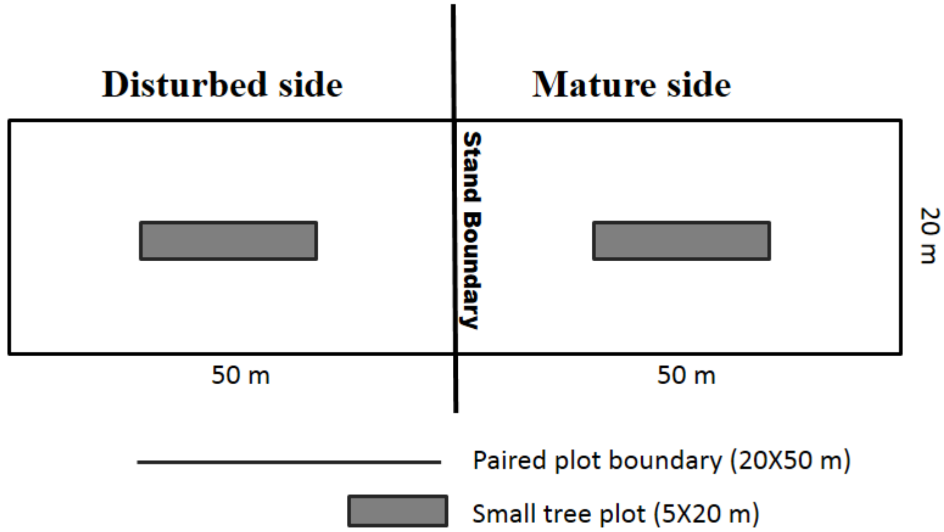

Figure 2.3 Plot design for forest structure sampling. Paired plots are 0.1 ha (20 m by 50 m) on each side of an edge. Small trees plots (0.01 ha) are located in the center each side of the paired plot.

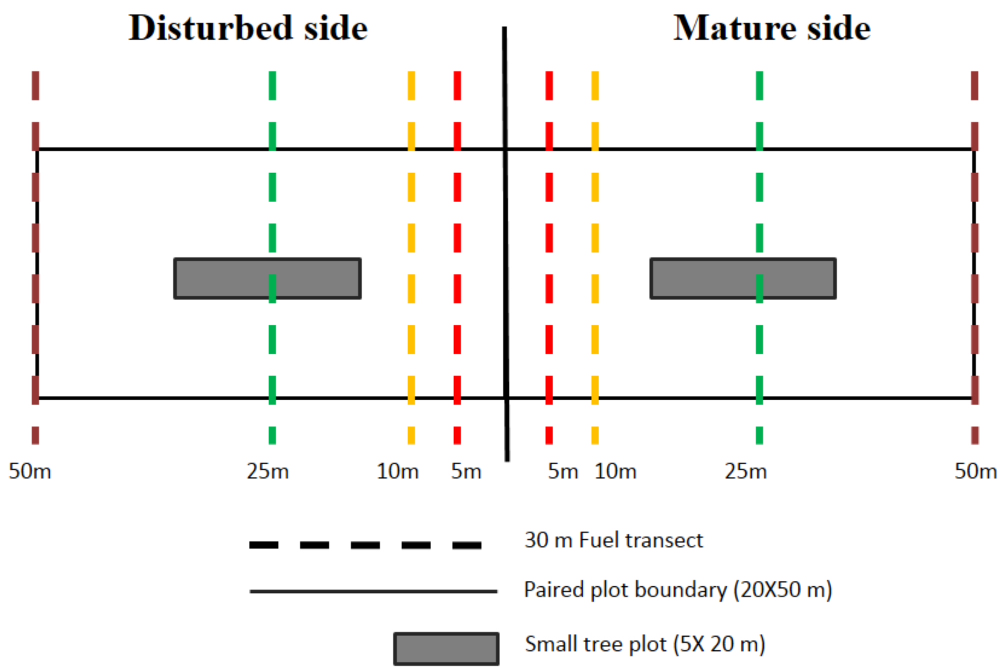

Figure 2.4 Plot design for fuel sampling. Four 30 m fuel transects were installed on both side of each edge.

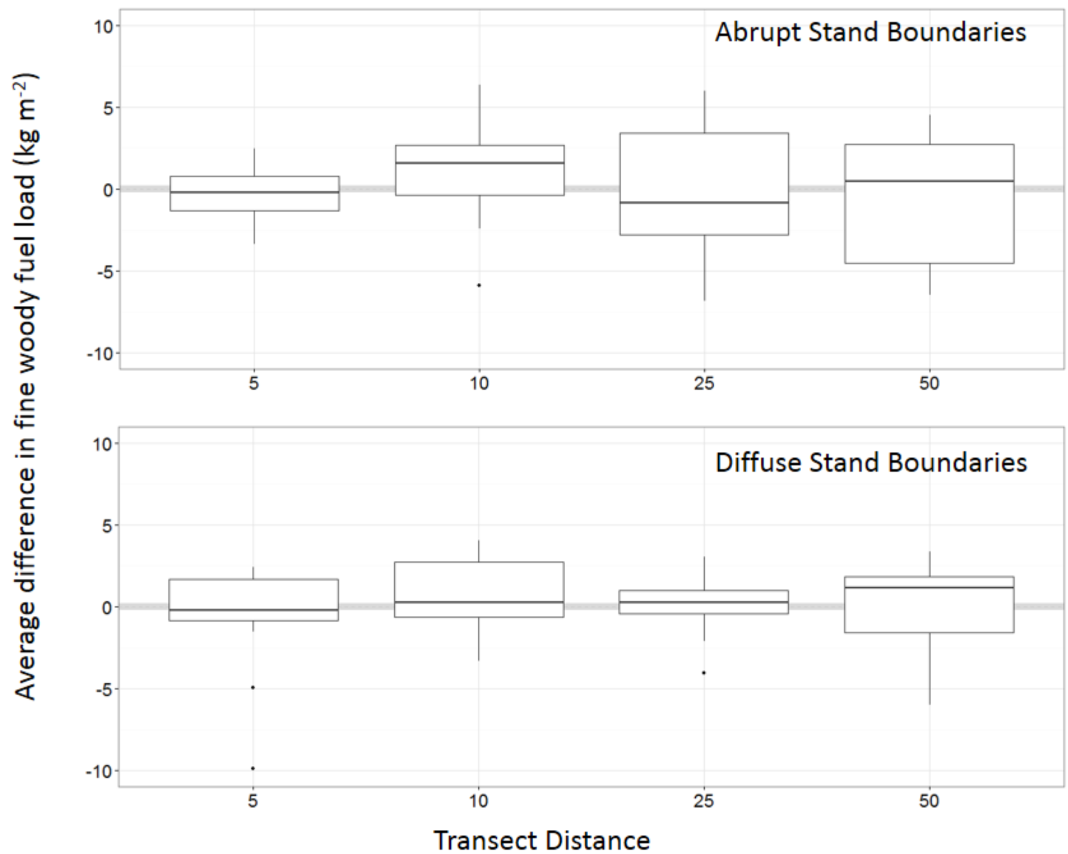

Figure 2.5 The difference in fine woody fuels across stand boundaries did not vary by edge type (abrupt or diffuse) or distance from edge. The grey bar indicated values that are not considered biologically significant. The bottom and the top of the box bound the 2nd and 3rd quartiles of the data (the middle 50%). The line inside the box is the median value. The whiskers above and below the box indicate data values that are less than 1.5 times the interquartile range more than the 3rd quartile or less than the 1st quartile. Dots represent data values that are more than 1.5 times greater than the 3rd quartile or less than 1.5 times the interquartile range below the 1st quartile.

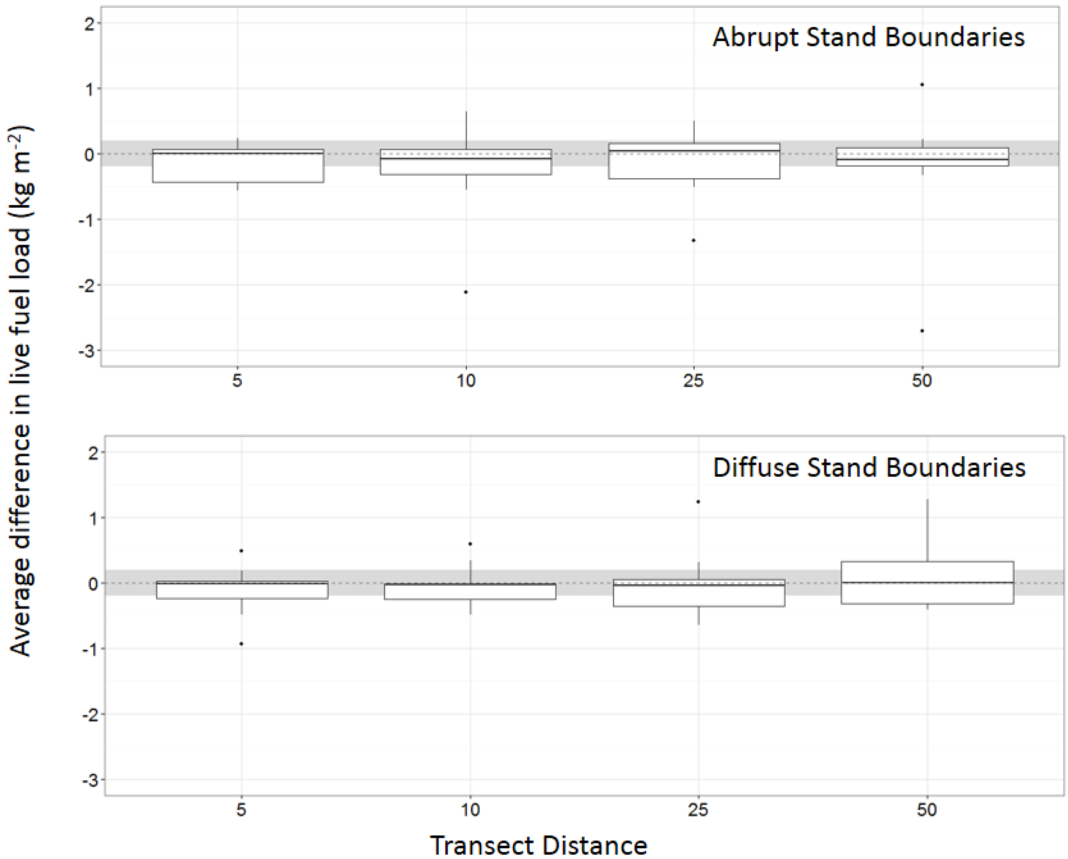

Figure 2.6 The difference in live fuel load across stand boundaries did not vary by edge type (abrupt or diffuse) or distance from edge. The grey bar indicated values that are not considered biologically significant. The bottom and the top of the box bound the 2nd and 3rd quartiles of the data (the middle 50%). The line inside the box is the median value. The whiskers above and below the box indicate data values that are less than 1.5 times the interquartile range more than the 3rd quartile or less than the 1st quartile. Dots represent data values that are more than 1.5 times greater than the 3rd quartile or less than 1.5 times the interquartile range below the 1st quartile

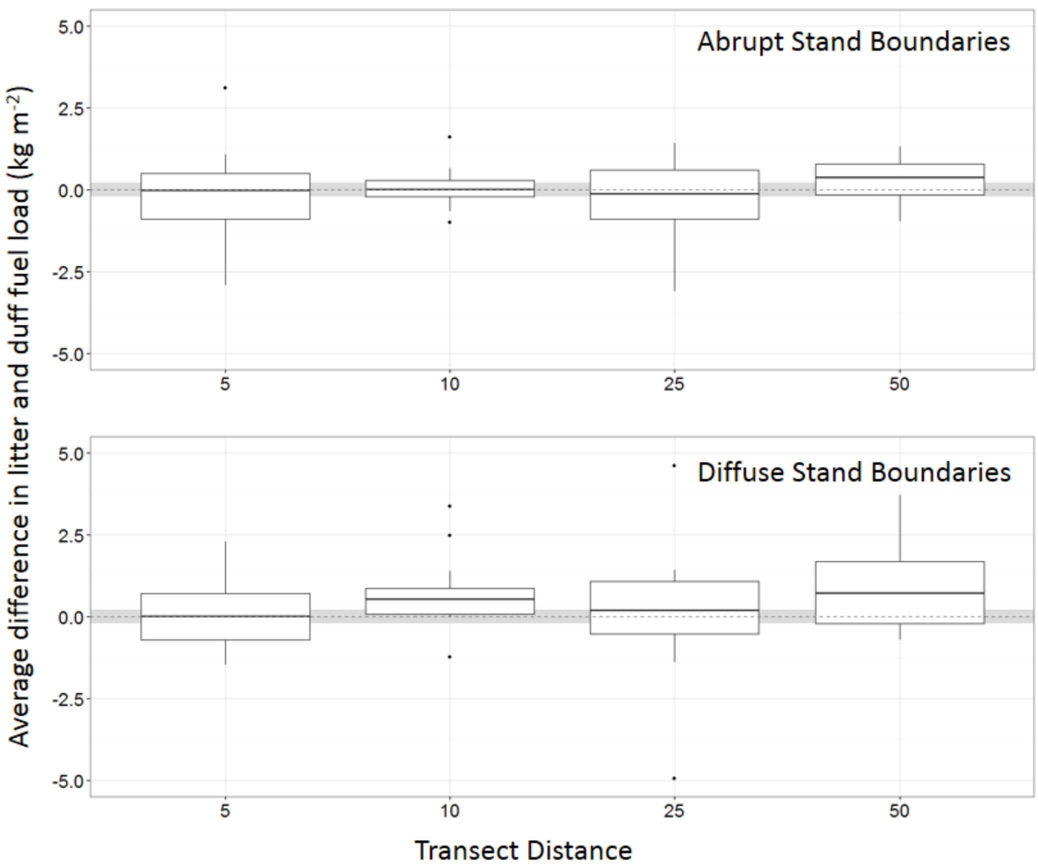

Figure 2.7 The difference in litter and duff fuel load across stand boundaries did not vary by edge type (abrupt or diffuse) or distance from edge. The grey bar indicated values that are not considered biologically significant. The bottom and the top of the box bound the 2nd and 3rd quartiles of the data (the middle 50%). The line inside the box is the median value. The whiskers above and below the box indicate data values that are less than 1.5 times the interquartile range more than the 3rd quartile or less than the 1st quartile. Dots represent data values that are more than 1.5 times greater than the 3rd quartile or less than 1.5 times the interquartile range below the 1st quartile

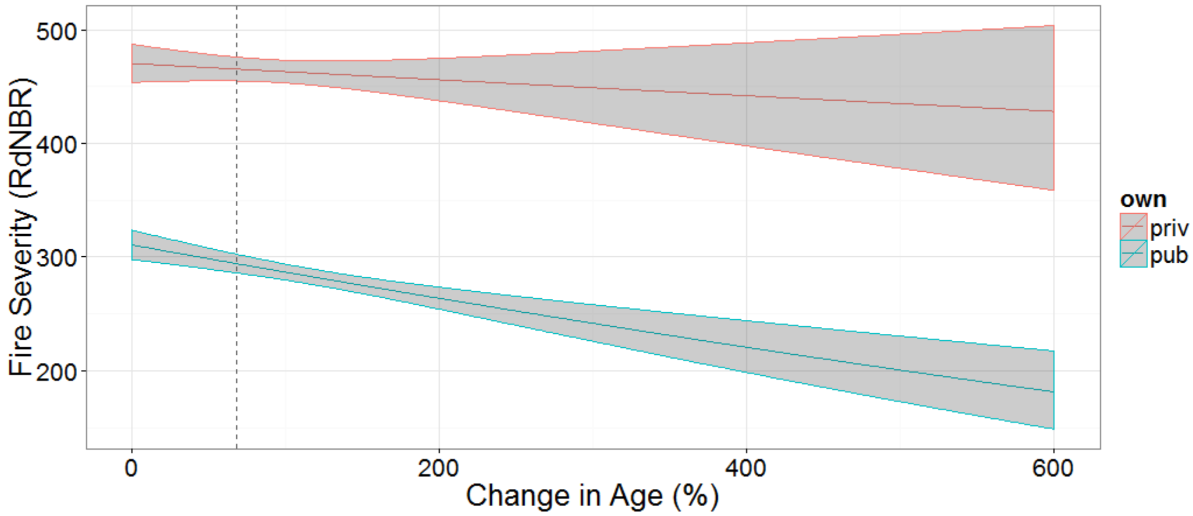

Figure 2.8. Change in RdNBR (disturbance severity) relative to the rate of change in age-structure across space for an average value of pre-fire NBR (pre-fire vegetation). As the rate of change in age-structure across space increases, disturbance severity remains high with little variation in privately owned (red), due to immediate post-fire salvage-logging. On publically owned land (blue), no salvage-logging occurred and there is a decrease in RdNBR as the rate of change in age-structure across space increases. The shaded areas represent the 95% confidence interval limits.

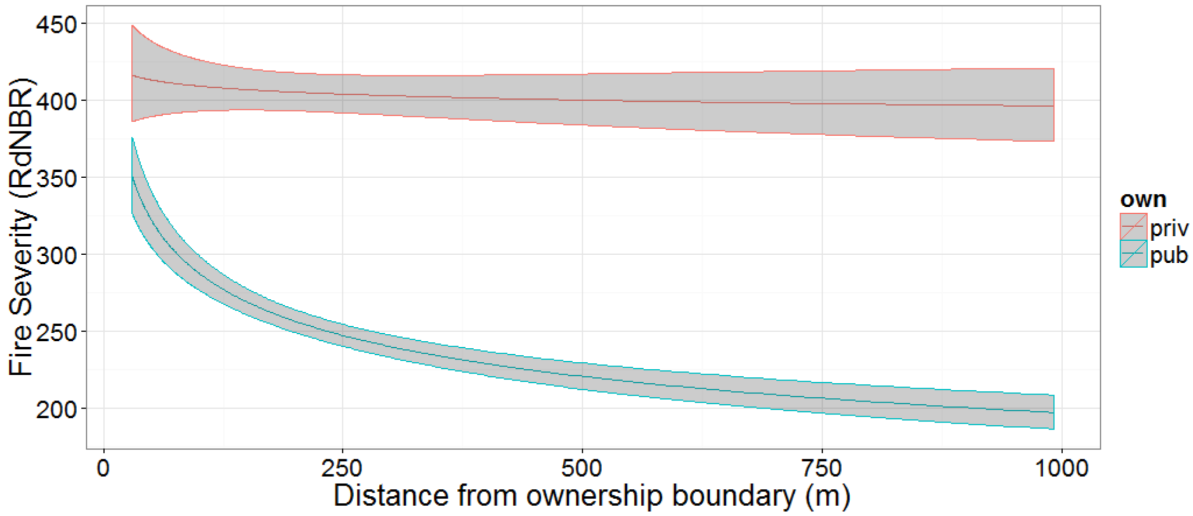

Figure 2.9. Change in RdNBR (disturbance severity) relative to distance from ownership boundary for an average value of pre-fire NBR (pre-fire vegetation condition). As distance from a boundary with publicly owned land increases, disturbance severity remains high with little variation in privately owned, due to immediate post-fire salvage-logging. On publically owned land (blue), no salvage-logging occurred and there is a decrease in RdNBR as distance to privately owned land increases. The decrease is greatest within 250 m of the ownership boundary. The shaded areas represent the 95% confidence interval limits.

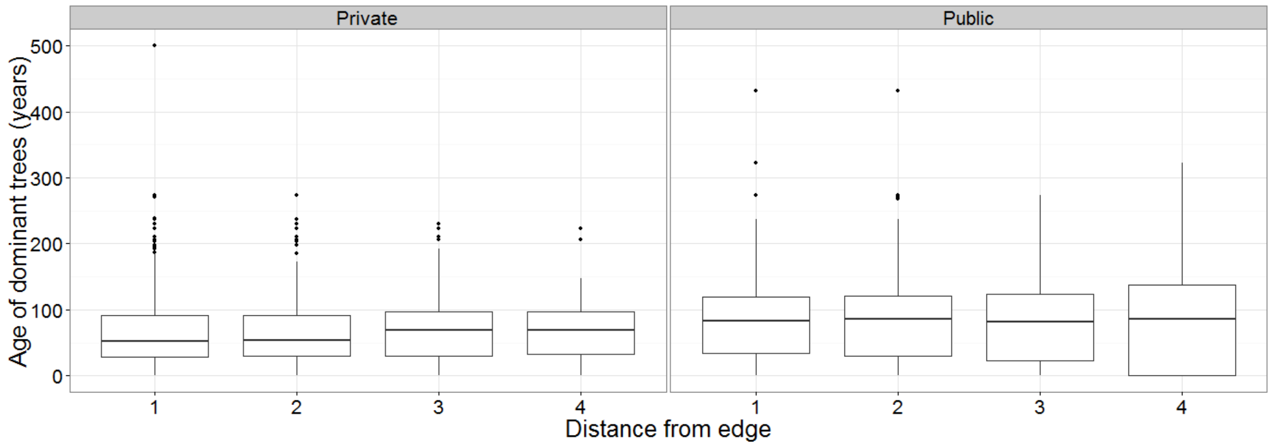

Figure 2.10 Pre-fire age structure at increasing distance from ownership edges on both private and public lands in the Elk Creek watershed (Jackson County, Oregon, USA) prior to the 2002 Timbered Rock fire. The distance classes are 1: 0-250 m, 2: 250-500 m, 3: 500-750 m, and 4: 750-1000 m. Ages are generally higher and more variable on publically owned land. The bottom and the top of the box bound the 2nd and 3rd quartiles of the data (the middle 50%). The line inside the box is the median value. The whiskers above and below the box indicate data values that are less than 1.5 times the interquartile range more than the 3rd quartile or less than the 1st quartile. Dots represent data values that are more than 1.5 times greater than the 3rd quartile or less than 1.5 times the interquartile range below the 1st quartile.

Tables

Table 2.1. Significance test, mean value, and 95% confidence intervals for the difference from the mature side to the disturbed side of abrupt and diffuse stand boundaries.

			abrupt		diffuse	
Structure Variable	**F(1,27)**	**p-value**	**mean difference**	**95% CI**	**mean difference**	**95% CI**
Basal area (m2/plot)	8.39	0.0073	3.9	0.8 to 4.5	1.3	-0.1 to 2.6
num trees/plot	0.02	0.8915	5	-50 to 57	1	-37 to 40
mean ht (m)	6.35	0.0179	5.6	0.9 to 8.5	0.9	-1.8 to 3.6
max ht (m)	16.1	0.0004	30.6	10.7 to 33.1	8.6	0.6 to 16.7
lcr	4.7	0.0391	-0.19	-0.19 to -0.01	-0.09	-0.15 to -0.02

Table 2.2. Results from MRPP group-wise comparison of structure groups

Group Comparison	T	A	p-value
AR vs AO	-9.81	0.17	<0.0001
AR vs DR	-1.82	0.03	0.059
DR vs DO	0.69	-0.01	0.7187

CONTRIBUTION OF AUTHORS

This paper was co-written by Emily Comfort with input from Darren Clark, Bob Anthony, and Matt Betts. The analysis was designed and implemented by Emily Comfort with guidance from Matt Betts and Bob Anthony. Telemetry data was collected by Darren Clark.

CHAPTER 3: NORTHERN SPOTTED OWL USE OF EDGES FOLLOWING THE MIXED-SEVERITY TIMBERED ROCK FIRE AND SALVAGE LOGGING, SOUTHERN OREGON, USA.

Abstract

While there is little debate as to the importance of late-seral, complex forest habitat for northern spotted owls (*Strix occidentalis caurina*), questions remain about the spatial configuration of this habitat, particularly in regions characterized by recent disturbance. Edges vary in structure based on the definition of the patches. They can grade from very abrupt edges, where patches are very different, to diffuse, where patches are similar. Past research has identified edges between spotted owl habitat and non-habitat as a potentially important habitat feature for spotted owls, however the broad definition of edge has led to some conflicting findings. I hypothesized that edges could be defined more precisely to tease out the individual effects of different kinds of edges. This research used telemetry data for 23 spotted owls following an 11,000 ha fire in southern Oregon to explore a new approach for measuring edges created by mixed-severity fire. My approach quantified the steepness of gradients directly by measuring the ‘slope’ of change in fire severity. I found that spotted owls had a strong negative association with hard edge after accounting for habitat suitability, disturbance severity, and amount of diffuse edge. Spotted owls were 0.710 (95% CI: 0.6808 to 0.7406) less likely to be present for every 0.09 ha increase in the amount of hard edge within 3.2 ha of a location. On the other hand, spotted owls generally had a positive, but weak association with the amount of diffuse edge. After accounting for other factors, probability of spotted owl use increases 1.001 (95% CI: 1.0007 to 1.0014) for every 0.09 ha increase in diffuse edge within 207 ha of a location. This resulted in an overall 15% increase in probability of occurrence across the range of diffuse edge we observed. The results of this study also confirm that previous models of spotted owl habitat suitability are strong predictors of spotted owl occurrence. Disturbance severity was negatively associated with spotted owl occurrence. Taken together, my results indicate that spotted owls select small

dispersed patches of high-severity disturbance in a matrix of low- and moderate-severity disturbance that maintains large patches of high-suitability habitat- a condition which is in line with historic fire regimes. My results support the use of metrics that have the capacity to differentiate between structurally diverse edges.

Introduction

While there is little debate as to the importance of late-seral, complex forest habitat for northern spotted owls (*Strix occidentalis caurina*) (US Fish and Wildlife Service 2011), questions remain about the importance of habitat configuration in the landscape matrix (Hunter et al. 1995, Ripple et al. 1997) and spotted owl use of post-disturbance environments and less complex forest (Irwin et al. 2000, Bond et al. 2002, Irwin et al. *In press*). Additionally, landscape ecologists are increasingly recognizing the importance of a gradient approach to landscape analysis versus the classic patch-mosaic model (Fischer et al. 2004, McGarigal and Cushman 2005, Cushman et al. 2010). These gaps in knowledge are important in southern Oregon dry forests, where spotted owls most likely were adapted to a mixed-severity fire regime prior to Euro-American settlement (Taylor and Skinner 1998, 2003). The changes in landscape pattern due to a century of logging, grazing, and fire-suppression have changed the configuration and structure of spotted owl habitat.

Northern spotted owl use of forest edges, especially those created by wildfire and management is not well understood. Several studies have found that a “hard edge” characteristic may be important to spotted owl habitat selection. Hard edge is often measured as the length in kilometers between suitable and non-suitable habitat. Hard edge was positively correlated with home range and core area size of spotted owls following the Timbered Rock fire in the southern Cascades (Clark 2007) and negatively correlated with territory occupancy (Clark et al. 2013). This suggests that, at the home range scale, hard edge has a negative effect on spotted owls. However, in the same study, spotted owl telemetry locations were marginally closer to hard edge than randomly-located, available sites, which could suggest selection for hard edges at small scales. Hunter et al. (1995) compared both fragmentation and landscape heterogeneity at varying spatial scales among spotted owl nest sites, roosting sites, and random sites in northern California. They found lower fragmentation (less edge) in nesting and roosting sites than at random sites at an 800m radius spatial scale, but landscape heterogeneity was similar among all three types of sites. The authors suggest this may be due to the utility of having some open and early-seral habitat for foraging. Other studies have found the amount of hard edge to be positively associated with survival (Franklin et al. 2000) and productivity (Franklin et al. 2000, Olson et al. 2004). However, Dugger et al. (2005) did not find edge to be related to reproductive rate or survival in southern Oregon. The variability in these results could be due to 1) the wide variability in structure of a single definition of edge obscuring the effect of structurally dissimilar types of edges and/ or 2) the confounding of edge with habitat amount and/or 3) variability in local study areas.

Loss of suitable northern spotted owl habitat to high-severity wildfire is a concern in southern Oregon (Roloff and Haufler 1997, Irwin and Wigley 2005, Roloff et al. 2005, Roloff et al. 2012). Old-growth dry, mixed-conifer forests in this area are in shorter supply and at higher risk of stand-replacing fires than in the past due to land management practices, including livestock grazing, timber harvests, and fire suppression in the last century (Borman 2005). Spotted owls in southern Oregon are likely to be adapted to the historical mixed-severity fire regime, but a change in the patch size and configuration of fire and fire severity in more recent decades (Westerling et al. 2006) can have detrimental effects on successional pathways (Levin and Paine 1974). While some debate continues on the future potential for large-scale loss of spotted owl habitat to wildfire (Hanson et al. 2009, 2010, Spies et al. 2010), researchers continue to examine short-term biological response of spotted owls to different severities of wildfire (Bond et al. 2002, Clark 2007, Bond et al. 2009, Clark et al. 2011, 2013). Bond (2009) suggested that four years after a fire, California spotted owls foraged in mixed-conifer forests that experienced high-severity fire. Clark et al. (2013) found that territory occupancy by spotted owls declined in the three years following the Timbered Rock fire and that extinction probabilities increased in the three years following three fires in southern Oregon (Timbered Rock, Quartz, and Biscuit fire) as the combined areas of early seral forest, high-severity fire, and salvage logging increased within the nesting core area. These apparently conflicting results could suggest that the scale and landscape context of high-severity fire is important for spotted owls. However, to my knowledge, no studies have measured fire effects on habitat selection at multiple scales.

Previous work in landscape ecology has typically measured edges as distinct features on the landscape (Hargis et al. 1998). In most landscapes that are not anthropogenically disturbed, however, edges occur as gradients. Hard edges may be created by disturbance events such as high-severity fire or logging where the disturbance is adjacent to mature forest (Figure 4.1). More diffuse edges are also common where less severe disturbance occurs or as the hard edges age (Figure 4.1). These different kinds of edges may be used quite differently by spotted owls and other species. Approaches to incorporate the gradient nature of these categorical variables have not been well developed (McGarigal and Cushman 2005, Cushman et al. 2010). Additionally, it has been difficult to incorporate metrics that separate the effects of habitat edge from amount of habitat (Betts et al. 2006).

It is well known that patterns and processes can change depending on the temporal and spatial scale at which they are observed (Wiens 1989). In order to determine the correct scale of analysis for a given phenomenon, it is important to determine the appropriate scale at which the processes that drive them operate. However, scale is rarely treated as treatment variable in experimental ecology; instead, it is considered background variable that needs to be accounted for in statistical tests (Sandel and Smith 2009). I hypothesize that the patch size and configuration of fire severity within a fire perimeter has a significant influence on how spotted owls will use the landscape. High-severity fire edges that occur as small patches dispersed in larger low-severity fire patches, may open up growing space for shrubs or new cohorts of trees and improve habitat for small mammal prey (Sakai and Noon 1993, 1997), thereby increasing spotted owl use at these edges. High fire severity edges that occur adjacent to large patches of high-severity fire or salvage logging may also open up growing space for shrubs and new cohorts of trees, but are less proximal to mature forest conditions, thereby reducing spotted owl use. Additionally areas that were intensively managed following the fire were most likely treated with herbicides in order to reduce the shrub competition for crop trees, which may decrease prey use (Sakai and Noon 1997). Spotted owls may respond independently to all these factors, so it is important to determine the scale and configuration to which they respond.

This research used movement data of spotted owls following a large, mixed-severity disturbance event to determine use versus availability of hard and diffuse edges used by spotted owls. My objectives were 1) to test a new metrics that may better describe the gradient nature of edges created by mixed severity fire and 2) to determine the scale at which spotted owl select different types of disturbance-created edges in the 2002 Timbered Rock Fire.

Methods

Study Area

The 2002 Timbered Rock Fire burned 11,000 ha on both federally managed land and private industry land (Figure 4.2). It is located in the Southern Cascades and Siskiyou Foothills ecoregions (McNab and Avers 1994). Vegetation is mixed conifer and mixed evergreen. The elevation range is 526 m to 1443 m. Precipitation ranges from 127 to 381 cm yr^{-1} and occurs mostly during the winter and spring. Temperatures range from 0° C to 40° C. Salvage logging occurred immediately following the fire on private industry land within the fire perimeter and there was no salvage logging on public lands. Differences in fire severity and post-fire management resulted in a mosaic of disturbance severities that disturbed seven northern spotted owl home ranges directly. An additional five spotted owl home ranges were located immediately adjacent to the fire boundary.

Spotted owl telemetry data

Locations of 23 spotted owls were collected between September 2002 and May 2006 with the use of ratio-telemetry methods. Each owl was monitored for at least 12 months unless the owl died or the transmitter failed. Five nocturnal locations and two diurnal locations were recorded for each owl every two weeks. Average telemetry error was 136m and locations were vectored when error polygons > 2 ha were obtained when triangulating a location. A total of 2,774 locations were recorded for the 23 spotted owls. For more details on telemetry data collection see Clark (2007).

Measuring edges as gradients across the landscape

Fire severity edges

Because edges grade from hard to diffuse, I used the local-scale (30 m pixel) spatial difference in fire severity to characterize the rate of change in fire severity at the 30 m scale. I acquired fire severity maps from Monitoring Trends in Burn Severity (MTBS) (Eidenshink et al. 2007) that depict the relative difference in normalized burn ratio (RdNBR); this index uses near infrared band 4 and short wave infrared band 7 from Landsat data to estimate fire severity from before and after the Timbered Rock fire to define fire created edges (Figure 4.2). Because salvage logging occurred immediately following the fire on private land in the study area, RdNBR cannot distinguish between fire severity and salvage logging severity. Nevertheless, the combined disturbance of salvage logging and fire severity resulted in higher and more spatially aggregated values of RdNBR on private land than on federally managed land (Chapter 2). The difference in patch size and severity resulted in a diversity of disturbance-created edges on the landscape, which is the landscape structure in which I was interested. I used the rate of change in RdNBR for a pixel and the adjacent pixels to calculate a percent "slope" map of disturbance severity, using the slope calculator in ArcGIS 10 (SLFS). The slope value is a rate of change index, similar to slope of elevation maps. Higher values indicate a larger difference in fire/salvage logging severity over the same spatial scale (similar to a steeper slope in an elevation map) (Figure 4.3).

Hard and diffuse edges

Similar SLFS values represent different kinds of fire severity edge on the landscape in the same way that a flat elevation slope can occur either on the top of a ridge or in a valley (same slope, different elevations). For this study, I wanted to differentiate between high SLFS values that occur in either high- or low- severity fire from high values that occur in moderate severity fire. These locations represent the areas with hard edges. High values in areas with moderate severity fire would not represent extreme differences, but would fit into my definition of diffuse edges, as would more moderate values of SLFS. Locations with low values of SLFS occur either in areas with contiguous low-severity fire or contiguous high-severity fire and therefore are not quantified as edge, but interior high-, moderate- or low- severity fire. I mapped edges into one of three categories, hard, diffuse, or non-edge, based on SLFS and RdNBR (Table 3.1). This process is similar to using an elevation and slope of an elevation map to identify various slope classes (e.g. ridges at high elevation). See Appendix 4 for more details. I conducted a sensitivity analysis of my thresholds for SLFS to explore how sensitive my results were to specific thresholds (results reported in Appendix 4).

Spatial scale

Finally, I wanted to examine edges at different scales across the landscape. I measured the amount (number of cells) of both hard and diffuse edge at six different spatial scales (0.8 ha, 3.2 ha, 12.9 ha, 51.8 ha, 207 ha, and 829 ha) around spotted owl telemetry locations and randomly selected available locations. The spatial scales correspond to different levels of use from foraging to core areas and home ranges. The different values gave measures of "edginess" at different spatial grains. If spotted owl selected for edginess at small extents, it would imply that stand-level disturbance effects (i.e. fire/salvage severity) are the structures that spotted owls selected. If spotted owls selected for edginess at larger spatial extents it would imply that landscape –level disturbance effects (i.e. the patchwork of fire/salvage severity within the fire) are the structures that spotted owls choose.

Statistical Analysis

I explored third-order habitat selection of spotted owls (Johnson 1980) following the Timbered Rock fire and subsequent salvage logging. I used a logistic mixed-effect regression model that compared 'used' versus 'available' spotted owl locations (Manly et al. 1992, Jones 2001) to identify habitat features that 23 spotted owls used more often than we would expect given their distribution and extent within each spotted owl's home range. "Used" sites were spotted owl telemetry locations and "available" sites were five randomly selected locations within an individual spotted owl's home range (e.g. if a spotted owl had 137 telemetry locations, I generated 685 random locations within its home range to define available sites). I generated five random sites for every spotted owl telemetry site nested within individual spotted owl home ranges (Warton and Shepherd 2010) in order to fully represent the range of available habitat conditions. Home ranges were delineated by a 100% minimum convex polygon around all telemetry locations for an individual spotted owl. The maximum distance an individual spotted owl travelled between consecutive telemetry locations (every 2-3 days or nights) ranged from 2 to 23 km, which made their entire home range available to them within this time frame.

In order to account for known important habitat features that I was not interested in exploring, I included habitat suitability as a fixed effect in all multivariate models. I measured habitat suitability from the Davis and Lint (2005) spotted owl habitat suitability index GIS layer. This map assigns habitat value for spotted owls from 1-100 (unsuitable to highly suitable) based on satellite-derived forest structure data (Ohmann and Gregory 2002) and spotted owl habitat models derived for specific regions from long term demography studies and multiple years of satellite data. I included disturbance severity (RdNBR) as a fixed effect as well because I thought it may be more important than or interact with my edge metrics. I included fixed effects of hard edge and diffuse edge. Hard edge is the sum of hard edge cells at a given spatial scale. Diffuse edge is the sum of diffuse edge cells at a given spatial scale. I included 'individual' as a random effect to account for lack of independence for successive observations of the same owl. This approach allowed for pairing of used vs. available locations for each individual owl.

I quantified each of my four habitat features at six different resolutions that roughly corresponded with different scales of spotted owl use (foraging, core use areas, and home ranges). For each spatial scale, I tested a univariate model that included only average habitat suitability against three multivariate models that represented different hypotheses about spotted owl habitat selection. H_1: spotted owls select locations based on average habitat suitability and average disturbance severity alone. (presence ~ habitat suitability + RdNBR); H_2: spotted owls select locations based on average habitat suitability. Amount of hard edge and amount of diffuse edge interact with average disturbance severity to further affect spotted owl selection. (presence ~ habitat suitability + RdNBR*hard edge + RdNBR*diffuse edge); H_3: spotted owls select locations based on average habitat suitability, average disturbance severity, the amount of hard edge and the amount of diffuse edge (presence ~ habitat suitability + RdNBR + hard edge + diffuse edge). I compared Bayesian information criteria (BIC) between models and chose the model with the lowest BIC. BIC ranks models based on fit to data, number of parameters, and sample size (Johnson and Omland 2004, Ward 2008).

Because my interest was to explore spotted owl use of edges, I then back-transformed the log odds of the fixed effects to determine the probability of spotted owl use. Fixed effects were considered statistically significant if the 95% confidence interval for the probability of use estimate did not include one.

Results

Spatial scale

Spotted owls generally used habitat characterized by higher habitat suitability, lower disturbance severity, lower amounts of hard edge and higher amounts of diffuse edge, but these results varied by scale of measurement (Figure 4.2). At three of the six spatial scales (829 ha, 12.9 ha, and 3.2 ha), models that included independent effects of habitat suitability, disturbance severity, hard edge, and diffuse edge were top-ranked or competitive with top-ranked models. At the 51.8 ha and 207 ha spatial scales, the best model included habitat suitability and disturbance severity interacting with both hard edge and diffuse edge. At the 0.8 ha extent, the model with only habitat suitability and disturbance severity had the best fit. Overall, the model that included average habitat suitability, average disturbance severity, total amount of hard edge, and total amount of diffuse edge at the 3.2 ha extent had the best fit to the data (BIC= 10,694.62).

Spotted Owl Habitat Suitability

Habitat suitability had a strong positive association with spotted owl habitat selection at all spatial extents. Habitat suitability values within the Timbered Rock fire boundary ranged from 0-81, but 96% of the values were between 1 and 62, suggesting that overall habitat quality was suboptimal. As expected, owl locations corresponded with the predicted distribution of habitat; for all spotted owl locations, average habitat suitability value was 38.8 (std dev 17.0). For all available locations, average habitat suitability value was 30.5 (std. dev 18.2). Spotted owl were 1.067 (95% CI: 1.063 to 1.072) times more likely to be present for every 1 unit increase in average habitat suitability within 12.9 ha of its location. In models that statistically controlled for RdNBR, hard edge, and diffuse edge, habitat suitability always had a significant positive association with spotted owl presence (Table 4.3a). In the best multivariate model (3.2 ha), spotted owls were 1.058 (95% CI: 1.054 to 1.062) times more likely to be present for every 1 unit increase in average habitat suitability within 3.2 ha of its location after accounting for the effects of average RdNBR, and total amount of hard and diffuse edge within 3.2 ha (Table 4.3a, Figure 4.4).

Disturbance severity

Disturbance severity had a strong negative association with spotted owl habitat selection at small and moderate spatial extents. The Timbered Rock fire and subsequent salvage logging resulted in a range of disturbance severity. Within the fire perimeter, 17% of the landscape was unburned or experienced very low-fire severity fire, 25% of the landscape burned at low-severity, 28% of the landscape burned at moderate severity, and 30% of the landscape burned at high-severity or was salvage-logged. Observed owl locations corresponded with the predicted distribution of disturbance severity; for all spotted owl locations, the mean RdNBR was 170 (std dev 231). For all available locations, the mean RdNBR was 240 (std dev 290). In the best multivariate model that statistically controlled for habitat suitability, hard edge, and diffuse edge, spotted owls were 0.999 (95% CI: 0.998 to 0.999) less likely to be present for every 1 unit increase in average RdNBR within 3.2 ha of the location (RdNBR range: -178 to 1013) (Table 4.3a, Figure 4.5).

Hard Edge

Hard edge was generally negatively associated with spotted owl habitat selection, but the relationship varied across spatial scales. Hard edge accounted for about 9% of the total area of the Timbered Rock fire area. Unexpectedly, in the best multivariate model (3.2 ha) that statistically controls for covariates, spotted owls were 1.299 (95% CI: 1.271 to 1.327) more likely to be present for every 3% increase in the number of hard edge cells in 3.2 ha. Hard edge was rare on the landscape, and my result appeared to be driven by a large number of zero values for available locations. There is no hard edge within 3.2 ha for 62% of spotted owl locations and 95% of available locations, which suggests that spotted owl use sites with some amount of hard edge in relation to the amount available. In the interest of exploring this result, I ran the model with all the zero values removed (used locations = 1062, random locations= 681), Spotted owl were 0.725 (95% CI: 0.693 to 0.758) less likely to be present for every 3% increase in the amount of hard edge within 3.2 ha. This result suggest that that spotted owls use small grains of hard edge more than it is available, but use quickly declines as the amount of edge within 3.2 ha increases (Figure 4.6). Our sensitivity analysis determined that the threshold I used to define hard edge based on SLFS and fire severity did not change my results qualitatively (see Appendix 4 for more detail).

Diffuse Edge

The amount of diffuse edge generally has a small positive effect on spotted owl selection. Diffuse edge accounted for about 54% of the total area of the Timbered Rock fire area. In the overall "best" model that included all the variables at a spatial extent of 3.2 ha, diffuse edge did not have a significant effect on spotted owl selection. However, at spatial extents from 12.9 ha to 207 ha, diffuse edge had a significant positive effect on selection after accounting for covariates. At the 207 ha extents, there was a 15% increase in probability of occurrence across the range of number of diffuse edge cells (Figure 4.7). Our sensitivity analysis determined that the threshold I used to define hard edge based on SLFS and fire severity did not significantly change my results (see Appendix 4 for more detail).

Discussion

Spotted owl use of disturbance-created landscapes

Our results provided strong support for the importance of disturbance severity (RdNBR) and edges created by gradients in disturbance severity to spotted owl selection. The strongest predictor of spotted owl presence following the 2002 Timbered Rock fire was habitat suitability, but disturbance severity, hard edge, and diffuse edge all increase the explanatory power of my models.

Spotted owls avoided large, contiguous patches of high-severity disturbance in my study. Previous studies of California spotted owls the Sierra Nevada mountains generally report that site occupancy is not impacted by high-severity fire (Lee et al. 2012) and that spotted owl forage in high-severity fire patches, but avoid high-severity fire for roosting (Bond et al. 2009). However, spotted owls in this study area had greater local extinction probabilities and a larger reductions in site occupancy than a nearby mature site that did not experience fire and salvage-logging (Clark et al. 2013). My results suggest that although some spotted owls with home ranges that overlapped the Timbered Rock fire included large proportions of high severity fire (range 4-39%), within those ranges they preferentially used areas of lower severity disturbance.

Spotted owls did not avoid hard edges within their home ranges following the 2002 Timbered Rock fire and subsequent salvage-logging, but they did select hard edge at fine scales. Spotted owls selected small scales (<0.8 ha) of hard edge within their home range more frequently than they are available, but as the scale increased (3.2 ha to 829 ha) and became more aggregated (increase in density of hard edge cells), spotted owls selected areas less frequently than they are available. This suggests that spotted owls can benefit from small patches of high severity fire that are surrounded by moderate and low-severity fire. The decreasing effect size at larger spatial scales could be a reflection of higher-order selection. At the home-range scale, spotted owls may be more likely to select for large patches of contiguous high suitability habitat interspersed with small patches (<0.8 ha) of high severity fire.

Spotted owls selected diffuse edges within their home range, but this effect was strongly mediated by disturbance severity. Spotted owl selected diffuse edges within intact highly suitable habitat that burned at low- to moderate-severity. After accounting for covariates, diffuse edges were most strongly associated with spotted owl selection at moderately large spatial scales (209 ha).

Diffuse edges are likely to be good habitat for woodrats (*Neotoma* spp) which are an important component of spotted owl diets in this region (Clark 2007). Woodrats were reported to occur at high densities in early seral (brushy/ sapling to pole- sized trees) and old-growth forests, but were absent in closed canopy stands with medium sized trees in Northern California mixed conifer/ hardwood sites (Sakai and Noon 1993). Shrub fields adjacent to old forests may increase the access to woodrats, who travel between the shrubs fields and openings in the old forest (Sakai and Noon 1997). Diffuse edges may also create better access for hunting small mammals, in general, while simultaneously providing adjacent closed canopy cover habitat. Bond et al. (2009) found that California spotted owls selected areas that burned at low- and moderate-severity for roosting and selected areas that burned at high-severity for foraging four years following the McNally fire in the Sierra Mountains. The authors suggested that increased prey availability (woodrats) is the explanation for the increased use. In my study, however, larger, more contiguous hard edges appear to have a negative impact on owl use. Hard edges at a large scale in this mixed-ownership landscape are mostly intensively managed edges which may lack the shrubby habitat that benefits woodrats. Instead, these hard edges may also expose spotted owls to predators and elements.

Mixed conifer forests in the Southern Cascades and Siskiyou foothill were historically characterized by high-frequency, mixed- severity fire regime (Agee 1991, Taylor and Skinner 1998, Beaty and Taylor 2001, Skinner 2003). Within this fire regime, much of the landscape would have burned at low- and moderate-severity leaving large conifer trees in open canopy forests and clearing out surface vegetation and some small and moderate sized trees. High severity fire created both small (<0.2 ha), ephemeral patches of open canopy (Scholl and Taylor 2010) and persistent shrub fields (Odion et al. 2004, Odion et al. 2010). While habitat loss due to large extents of high-severity fire is still a concern for long term spotted owl persistence (US Fish and

Wildlife Service 2011), my research suggest that spotted owls may respond favorably to high-severity edges (<0.8 ha) as long as these are embedded in a matrix of low- and moderate-severity fire. Spotted owls are likely adapted the patchy historical landscape in this region of their range. Spotted owls have a “bet-hedging” life history strategy (high adult survival and high annual variability in reproduction) suggesting a long-term adaptation to variability in environmental conditions (Franklin et al. 2000, Anthony et al. 2006). Spotted owl use of fire and salvage logging-created edges could be very different in the Oregon and Washington Coast Range where historic disturbance regimes were likely less frequent, more severe and were characterized by larger patch sizes (Agee 1996).

Gradient approach to measuring fire and salvage-logging- created edges

The quantitative approach I took to defining edges and modeling spotted owl habitat selection was useful in two respects. First, I was able to parse out the independent effects of two fundamentally different types of landscape ‘edge’. Second, I was able to separate the effect of habitat amount - a landscape composition variable- and edge -a landscape configuration variable (Fahrig 2003). Traditional views of habitat use view human-dominated landscapes as a patch mosaic of defined habitat or cover types defined as suitable or unsuitable (Haila 2002). Such approaches run the risk of neglecting important habitat features that do not fit into the patch-mosaic model of landscapes’ complex ecological structures such as edges (Fischer et al. 2004). Simplified definition of edges has resulted in conflicting and difficult to interpret responses of spotted owls to ‘edge’ (Hunter et al. 1995, Dugger et al. 2005, Clark et al. 2013). In my study, responses by spotted owls to ‘edge’ depended entirely upon whether or not such edges were classified as ‘hard’ or ‘diffuse’. Simplification also runs the risk of obscuring important patch effects. Schilling et al (2013) found that hard edge (the length of edges between suitable and non-suitable habitat) was the only variable in the best fitting model to explain home range size and core area size for spotted owls in a nearby study area. Our study underscores the importance of high suitability habitat as having the greatest effect on spotted owl selection.

At most spatial scales I considered, including edge variables in multivariate habitat selection models improved model fit. The diffuse edge variable performed more weakly in explaining spotted owl selection than the hard edge variable. Over 50% of the fire was defined as diffuse edge and there were correlations between disturbance severity and diffuse edge >0.7 at some spatial scales. While my sensitivity analysis found similar relationships to habitat selection for both a more restrictive definition of diffuse edge and a more generalized definition, the definition of diffuse edge I used in my final models could use further refinement in future analyses. Because disturbance severity, as it is defined by RdNBR, is a function of both what was there prior to the fire and what remained following the fire, it depicts fire effects more as a process than a structure. For defining diffuse edges particularly in forests rather than across the span of vegetation structures (grasslands, shrublands, savannas that would not be considered spotted owl habitat), if may be better to classify fore created edges by the rate of change in fire severity and a measure of forest structure (canopy cover or tree density) rather than RdNBR.

Conclusions

Spotted owl use patterns following the 2002 Timbered Rock fire examined in this study underscore the importance of retaining areas that have previously been predicted to be high-suitability spotted owl habitat (Franklin et al. 2000, Olson et al. 2004, Anthony et al. 2006). Specifically, my results suggest that spotted owls use small scales of high-severity fire in a matrix of moderate- and low-severity fire in high-suitability habitat. While "use" alone does not indicate fitness (Schlaepfer et al. 2002), under many conditions choices are adaptive. Ideally, use should be combined with measures of fitness to define suitability (Jones 2001). Clark (2011) did find lower survival rates for spotted owls that remained within the Timbered Rock fire perimeter and spotted owls that dispersed from the burned area compared to spotted owls in territories adjacent to the fire, which indicates that selection of fire-created edges occurred under suboptimal conditions and may not be beneficial to owls.

Recent research suggests that under various climate change scenarios, fire seasons will be longer and fire sizes will be larger (Westerling and Bryant 2008, Westerling et al. 2011) in the future. Additionally, loss of habitat to both past management practices and wildfire has diminished the amount of available, high-suitability habitat currently available for spotted owls (US Fish and Wildlife Service 2011). Our results support a management approach that takes a landscape-scale approach to maintaining and increasing the longevity of current high quality habitat by embedding high quality habitat in a more fire-resilient landscape. At the landscape-scale, applying frequent prescribed fire with some level of variable- retention harvests at small scales (>12.9 ha) could start to restore a more historic disturbance patch size and over longer periods of time promote the creation of diffuse edge at the scale at which I found spotted owl selecting for it. At very small scales (<0.8 ha), spotted owl selected for hard edge. Small patch-cut harvesting to reduce canopy continuity (and reduce canopy fire risk) may improve habitat quality in even-aged stands that are not currently high quality habitat. In the near term these cuts would create the hard edge at very small scales that spotted owl selected and as these cuts age, they will be transform into diffuse edge which can benefit spotted owls in the future. Unfortunately, this is not necessarily a feasible answer because in order to restore fire resilient structure and function to the landscape, cuts would have to have a small spatial grain and occur often in conjunction with prescribed fire to treat surface fuels, which would likely be expensive relative to any timber product it supplied. Additionally, my study was a case-study of spotted owl use after a single fire. Additional research should confirm these results prior to incorporating them into management plans

Fire does not have a single impact on landscape, rather it both burns variably across landscapes and impacts forest and landscape structure variably. Our edge metrics were able to parse out different disturbance-created impacts and model the landscape more variably that previous edge metrics that generally use a patch- matrix approach. While further refinement is needed, the method is promising.

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Figures

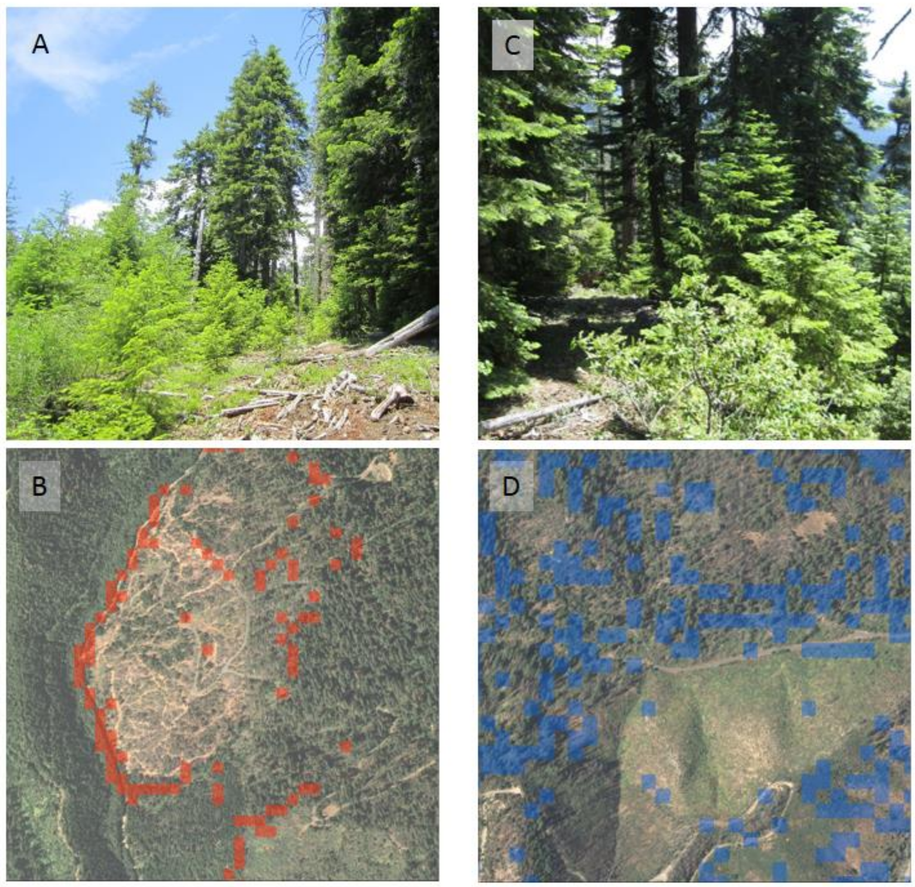

Figure 3.1 Ground-level (A) and landscape (B) view of a hard fire/ salvage logging created edge (colored red) compared to ground-level (C) and landscape (D) view of a diffuse fire/ salvage logging created edge (colored blue).

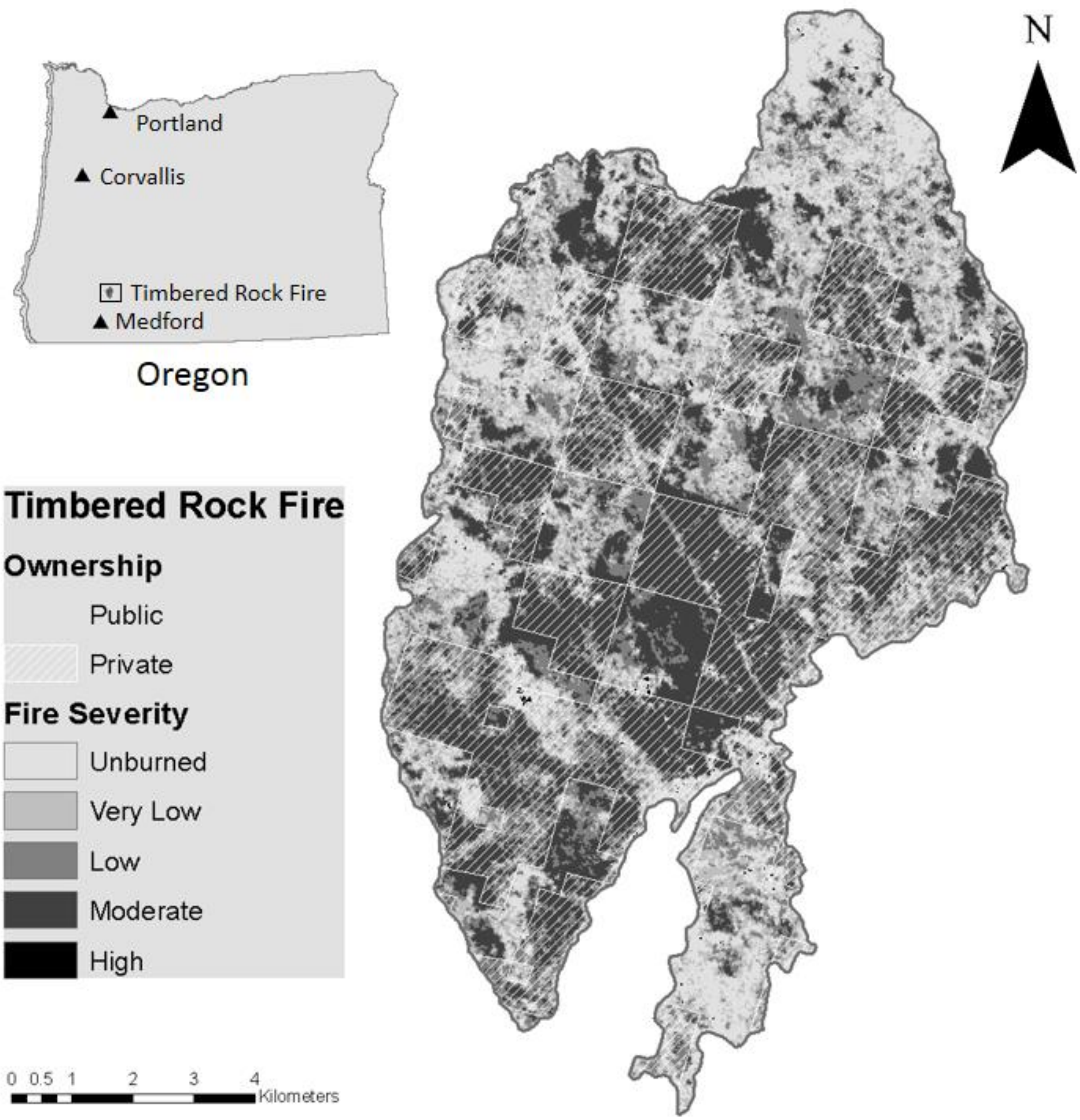

Figure 3.2 The 2002 Timbered Rock fire burned approximately 11,000 ha in mixed public and private ownership landscape in southern Oregon.

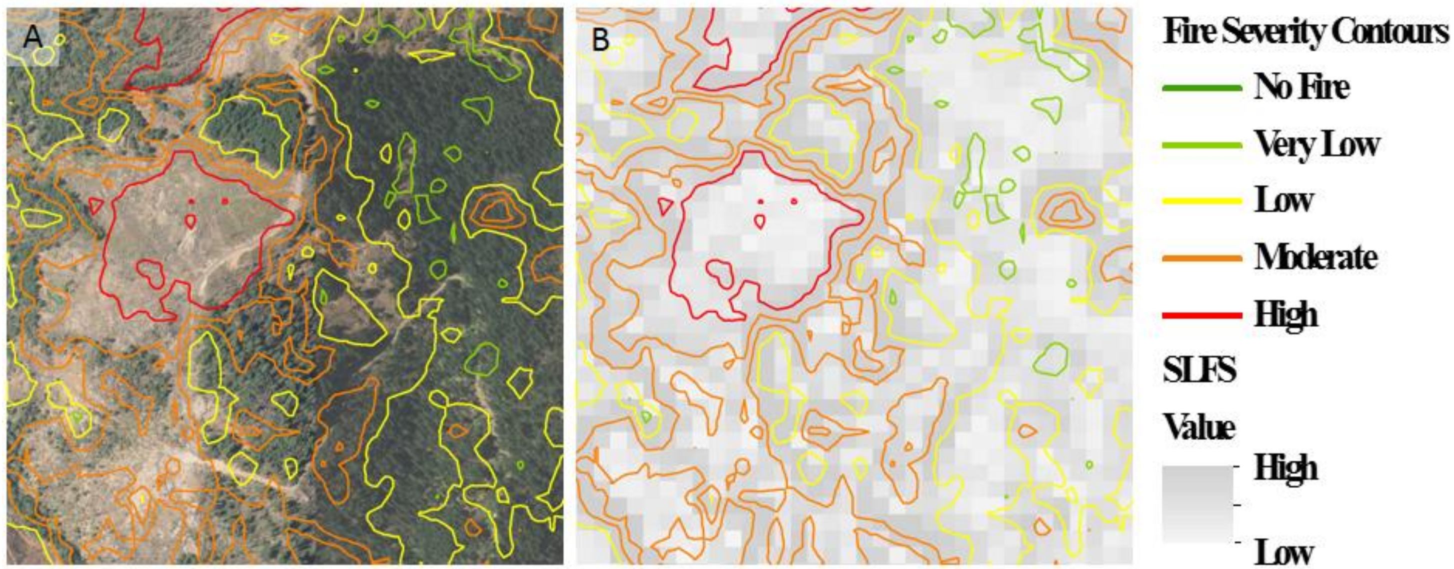

Figure 3.3 Map showing fire severity contours against (A) aerial image of salvage logged patch adjacent to mature forest and (B) SLFS values. Darker cells have high SLFS and are located in areas where the contour intervals of fire severity are close together. Light cells have low SLFS values and are located in areas with the contour intervals of fire severity are distant.

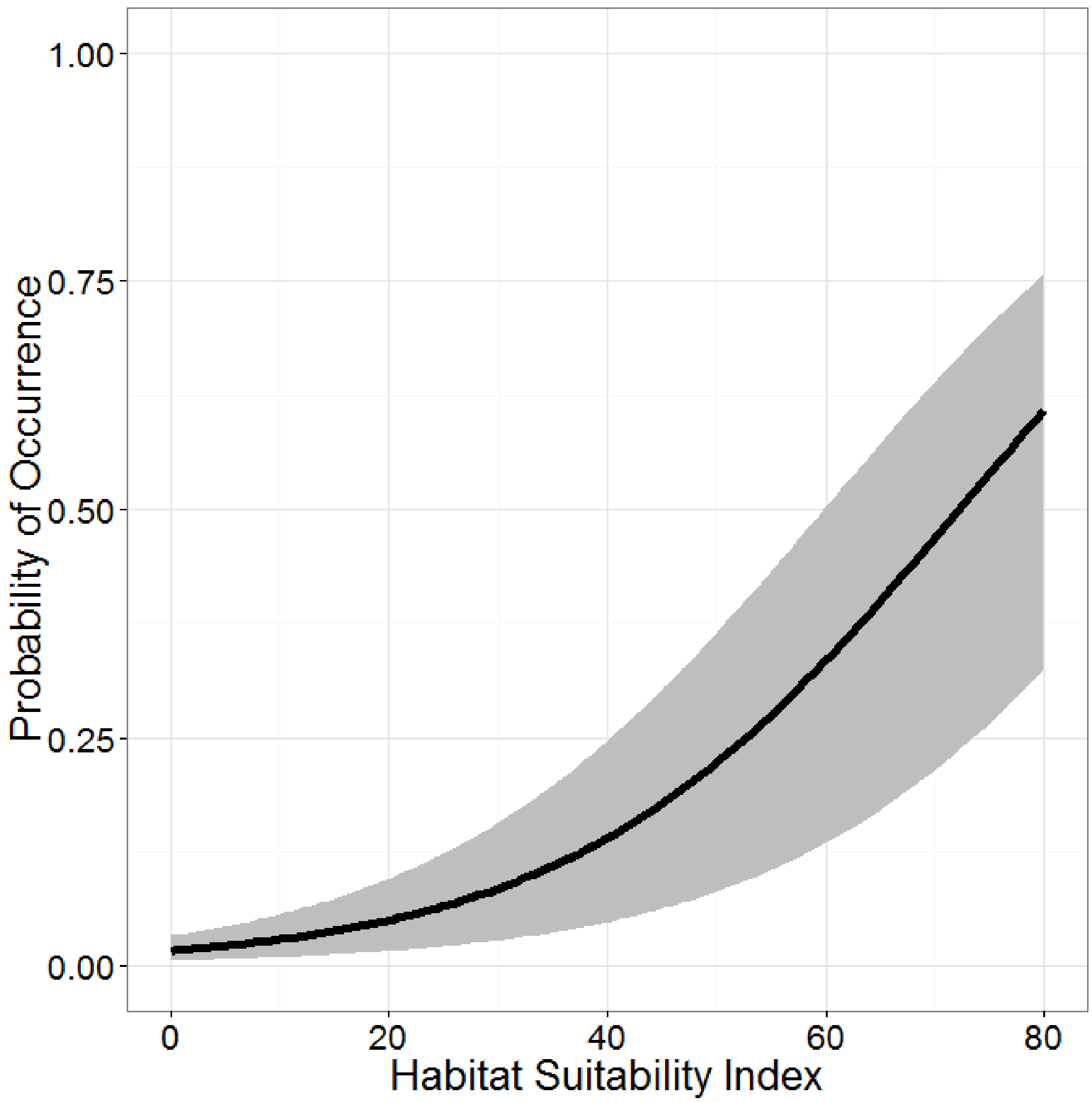

Figure 3.4 Probability of spotted owl occurrence increases as average habitat suitability within 3.2 ha of a location increases in a multivariate habitat selection model that accounts for fire/ salvage logging severity, hard edge, and diffuse edge.

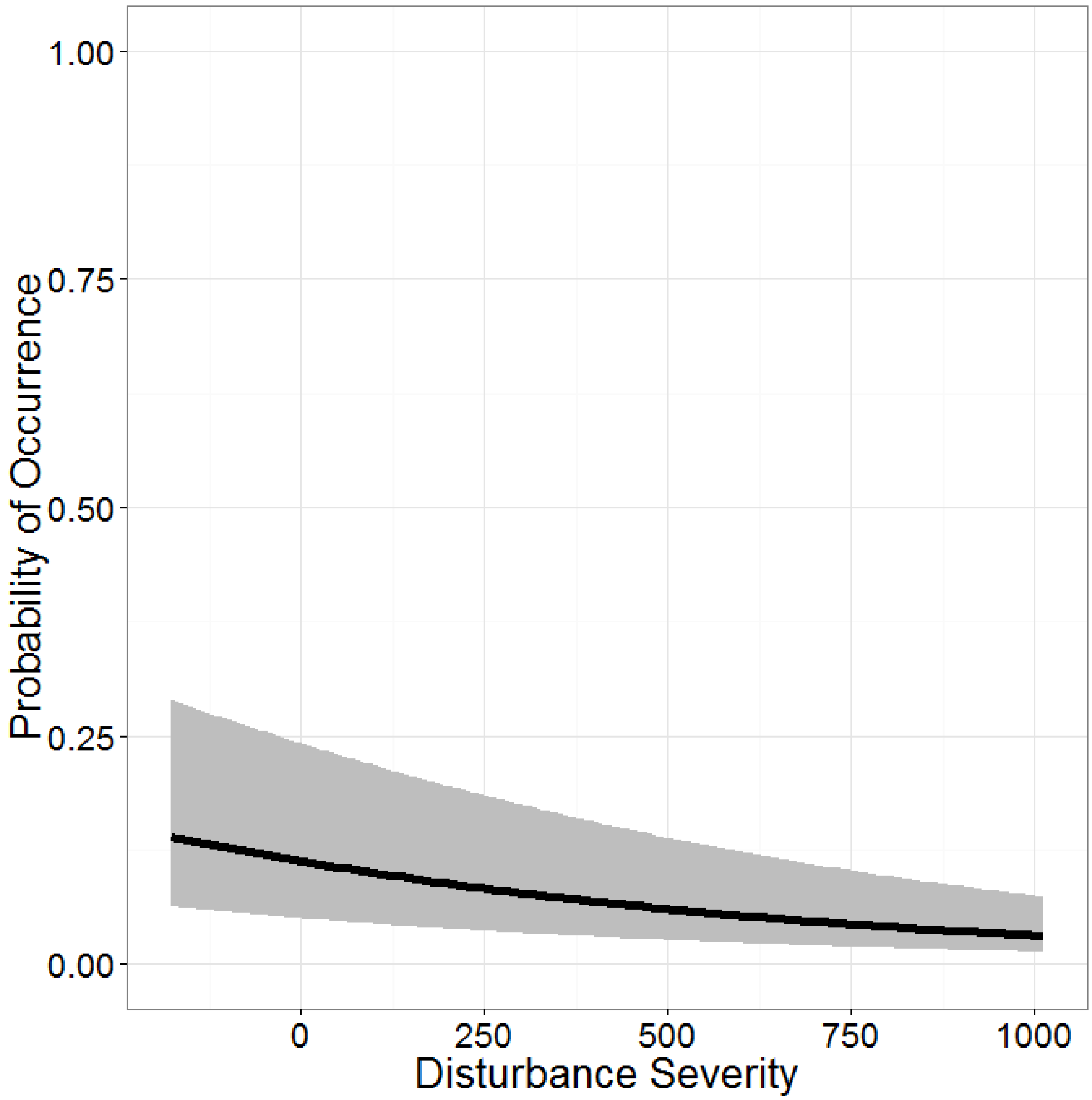

Figure 3.5 Probability of spotted owl occurrence decreases as average RdNBR within 3.2 ha of a location increases in a multivariate habitat selection model that accounts for habitat suitability, hard edge, and diffuse edge.

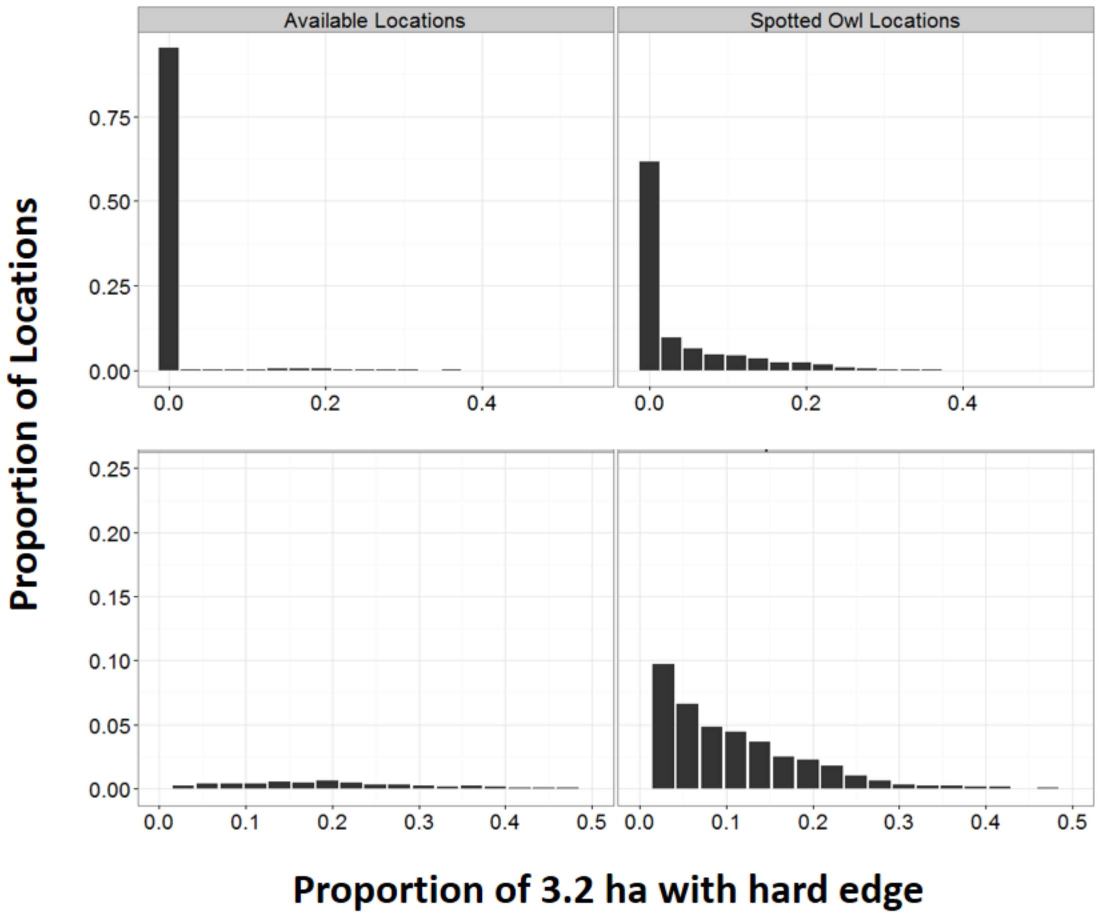

Figure 3.6 Spotted owls use locations that have no hard edge less than they are available within their home ranges, but they use areas with small patches of contiguous hard edge more than they are available and larger patches of contiguous hard edge less than they are available. The two rows show the same data, but the bottom row focuses on non-zero values of hard edge amount. Spotted owl locations are telemetry locations and available locations are random selected locations within an individual spotted owl's home range that represent the range of hard edge present in the landscape.

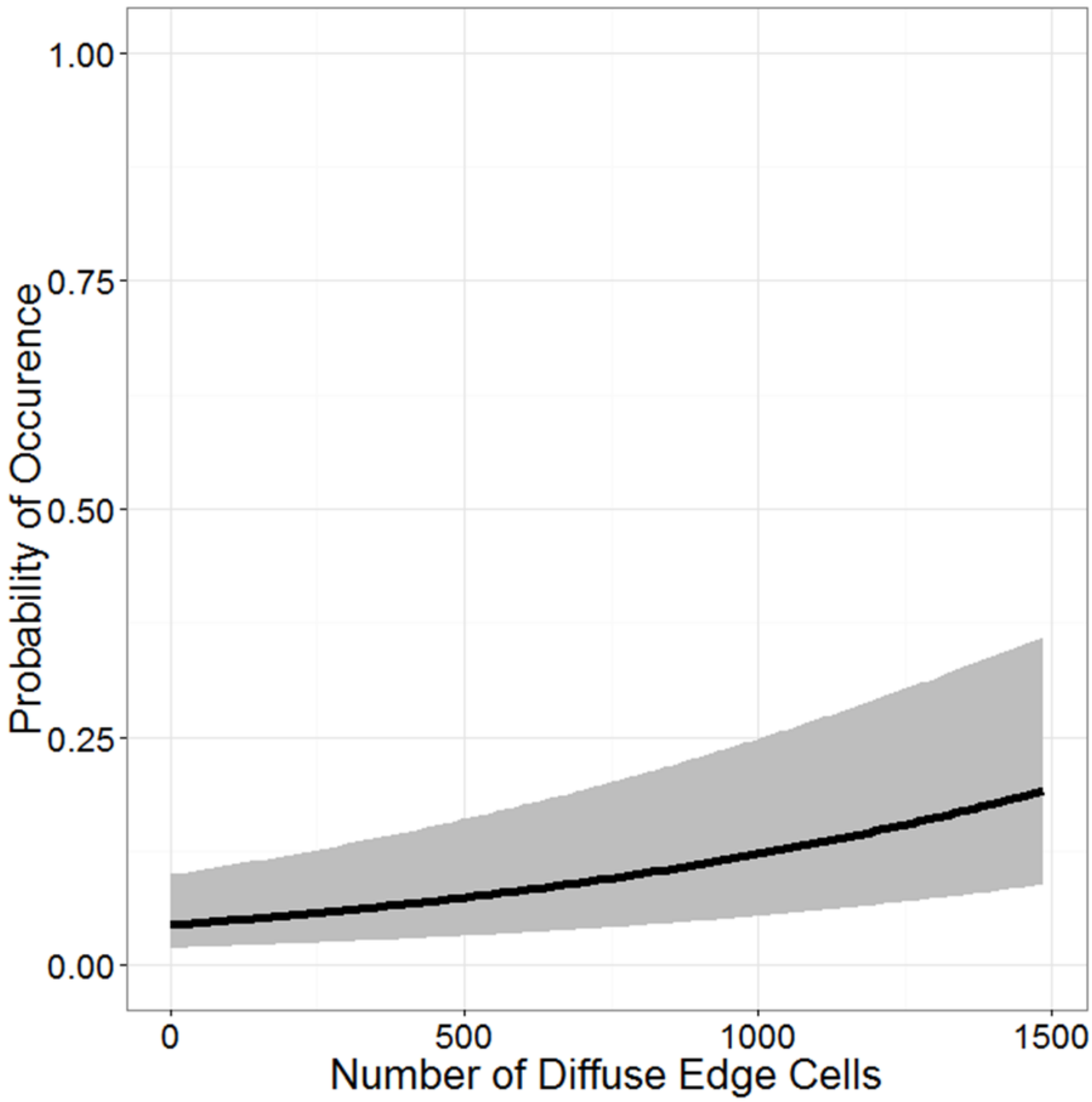

Figure 3.7 Probability of spotted owl occurrence increases as the sum of diffuse edge cells increases within 207 ha of a location increases in a multivariate habitat selection model that accounts for habitat suitability, fire/ salvage logging severity, and hard edge.

Tables

Table 3.1Thresholds used to assign edge status. RdNBR is a measure of fire/ salvage logging severity and SLFS is a measure of the rate of change (slope) of fire/ salvage logging severity.

	SLFS (%)		
RdNBR	209	209-534	>534
less than -150	Non-edge	Diffuse	Hard
- 150 – 150	Non-edge	Diffuse	Hard
150-350	No-edge	Diffuse	Hard
350-600	Non-edge	Diffuse	Diffuse
>600	Non-edge	Diffuse	Hard

Table 3.2. Ranked model selection results for post-fire habitat selection of spotted owls at six spatial scales used in this study (pres is the probability of occurrence, hab is habitat suitability index value, RdNBR is fire severity index, hard is the amount of hard edge and diffuse is the amount of diffuse edge).

Scale	model	K	BIC	ΔBIC
0.8 ha	pres~ hab + RdNBR	2	11299	0
0.8 ha	pres~ hab + RdNBR + hard + diffuse	4	11318	19
0.8 ha	pres~ hab + RdNBR*hard +RdNBR*diffuse	7	11321	22
0.8 ha	pres~ hab	1	11326	27
3.2ha	pres~ hab + RdNBR + hard + diffuse	4	10695	0
3.2ha	pres~ hab + RdNBR	2	10712	17
3.2ha	pres~ hab	1	11234	539
3.2ha	pres~ hab + RdNBR*hard +RdNBR*diffuse	7	11254	559
12.9 ha	pres~ hab + RdNBR + hard + diffuse	4	11067	0
12.9 ha	pres~ hab + RdNBR*hard +RdNBR*diffuse	7	11070	3
12.9 ha	pres~ hab + RdNBR	2	11118	51
12.9 ha	pres~ hab	1	11121	54
51.8 ha	pres~ hab + RdNBR*hard +RdNBR*diffuse	7	11117	0
51.8 ha	pres~ hab	1	11138	21
51.8 ha	pres~ hab + RdNBR	2	11148	31
51.8 ha	pres~ hab + RdNBR + hard + diffuse	4	11153	36
207 ha	pres~ hab + RdNBR*hard +RdNBR*diffuse	7	11336	0
207 ha	pres~ hab + RdNBR + hard + diffuse	4	11354	18
207 ha	pres~ hab + RdNBR	2	11504	168
207 ha	pres~ hab	1	11551	215
829 ha	pres~ hab + RdNBR + hard + diffuse	4	11872	0
829 ha	pres~ hab + RdNBR*hard +RdNBR*diffuse	7	11872	0
829 ha	pres~ hab + RdNBR	2	11902	30
829 ha	pres~ hab	1	12035	163

Table 3.3a. Results of multivariate model of spotted owl habitat selection as a function of average habitat suitability (hab), average fire/ salvage logging severity (RdNBR), amount of hard edge (hard), and amount of diffuse edge (diffuse) at size different scales from spotted owl and randomly selected available locations.

Model	Parameter	Estimate	SE	p-value	Odds	95%C.I. Odds Ratio
3.2 ha	Intercept	-2.301	0.475	<0.001	NA	NA
	Habitat Suitability Index	0.056	0.002	<0.001	1.058	1.054- 1.062
	Fire Severity Index	-0.001	<0.001	<0.001	0.999	0.998- 0.999
	Amount of Hard Edge	0.261	0.011	<0.001	1.299	1.271-1.327
	Amount of Diffuse Edge	0.003	0.005	0.575	1.003	0.994-1.012
12.9 ha	Intercept	-2.479	0.494	<0.001	NA	NA
	Habitat Suitability Index	0.061	0.002	<0.001	1.063	1.059- 1.068
	Fire Severity Index	-0.001	<0.001	<0.001	0.999	0.999-1.000
	Amount of Hard Edge	-0.032	0.004	<0.001	0.969	0.961-0.977
	Amount of Diffuse Edge	0.009	0.002	<0.001	1.009	1.006-1.012
51.8 ha	Intercept	-3.01	0.516	<0.001	NA	NA
	Habitat Suitability Index	0.078	0.003	<0.001	1.081	1.075- 1.087
	Fire Severity Index	-0.006	0.001	<0.001	0.994	0.992- 0.996
	Amount of Hard Edge	0.015	0.004	<0.001	1.015	1.007-1.021
	Amount of Diffuse Edge	-0.002	0.001	0.027	0.998	0.997- 1.000
	Fire Severity*Hard Edge	<0.001	<0.001	<0.001	1.000	1.000- 1.000
	Fire Severity*Diffuse Edge	>-0.001	<0.001	<0.001	0.999	-0.999-0.999
207 ha	Intercept	-4.19	0.542	<0.001	NA	NA
	Habitat Suitability Index	0.101	0.004	<0.001	1.081	1.075- 1.087
	Fire Severity Index	0.002	0.001	0.11	0.998	0.992- 0.996
	Amount of Hard Edge	-0.02	0.002	<0.001	1.014	1.008- 1.022
	Amount of Diffuse Edge	0.003	<0.001	<0.001	0.998	0.996- 1.000
	Fire Severity*Hard Edge	>-0.0001	<0.001	<0.001	1.000	1.000- 1.000
	Fire Severity*Diffuse Edge	<0.0001	<0.001	<0.001	1.000	1.000- 1.000
829 ha	Intercept	-4.15	0.054	<0.001	NA	NA
	Habitat suitability index	0.107	0.006	<0.001	1.113	1.100- 1.126
	Fire severity index	0.005	<0.001	<0.001	1.005	1.004-1.006
	Amount of hard edge	-0.002	<0.001	<0.001	0.998	0.998-0.999
	Amount of Diffuse edge	<0.001	<0.001	0.247	1.000	1.000-1.000

Table 3.2b. BIC scores for multivariate models.

Scale (ha)	BIC	ΔBIC
0.8	11318	623
3.2	10695	0
12.9	11067	372
51.8	11138	443
207	11353	658
829	11872	1177

CHAPTER 4: CONCLUSIONS

Dry-forest ecology and management in the Pacific Northwest are complicated. The structure and function of these forests has changed in the last century and a half, since Euro-American settlement (Heyerdahl et al. 2001, Hessburg et al. 2005) due to development and changes in land use, including fire suppression. The resilience of the current structure and function to climate change, future large fires, and other disturbances is questioned (Hanson et al. 2009, Littell et al. 2009, Spies et al. 2010). Future fire is a concern for threatened and endangered species, particularly the northern spotted owl (US Fish and Wildlife Service 2011).

Landscape management goals on federally-managed lands in the dry-forest region of southwest Oregon include increasing the resistance and resilience of forests to future disturbances by restoring a density and tree species composition more similar to pre-Euro-American settlement (Reilly 2012). Federal agencies are also constrained in their management practices by legislation and policies that range in time and scope from the O&C act of 1937, which mandated that BLM O& C land should be managed for a sustainable yield of timber, to the Endangered Species Act of 1973, which compels managers to preserve habitat associated with threatened and endangered species at the expense of other services. Often the mandates from these legislation are apparently conflicting. The Northwest Forest Plan intended to use a region-wide landscape ecology approach to bridge the goals of the different mandates, but it has not been implemented as intended.

Northern spotted owls are (*Strix occidentalis caurina)* have become an umbrella species for old-forest conservation. There is a perceived conflict in dry-mixed-conifer forests between retaining beneficial NSO habitat and reducing fire risk (Roloff et al. 2005, Ager et al. 2007, Roloff et al. 2012). Environmentalists are concerned 20th century logging, prior to implementation of the Northwest Forest Plan, has reduced the amount of Old-growth forest. The forest products industry is concerned that timber reserves on public land are being lost to wildfire and overall reductions in intensive management on public lands is diminishing the capacity of the timber industry to support jobs and provide timber. Increasingly, management aimed at reducing short-term fire risk through small-scale fuels reduction treatments in the wildland-urban interface are the only widely accepted management practices.

The central thesis of my dissertation is the exploration of landscape ecology theories in dry-forest applications. In the first chapter, I reviewed dry-forest historical ecology related to the southwestern Oregon and northern California Klamath region. This region is considered a hotspot of biological diversity due many factors including the complicated physiography and geology, climatic variability, and a long history of mixed-severity fire (Whittaker 1960). Both long and short term fire history studies suggest that over long reaches of time (1000's of years), fire regimes have changed in response to climate (Mohr et al. 2000, Whitlock et al. 2003) and over short periods of time (100s of years) the structure of the landscape (productivity) influences fire occurrence and spread (Heyerdahl et al. 2001, Taylor and Skinner 2003), while extreme weather events are associated with the highest severity fire in recent years (Weatherspoon and Skinner 1995, Thompson and Spies 2009). Since Euro-American settlement, the landscape has become both more fragmented by development, commercial forestry, and other land-use changes and more uniform through fire suppression. Fragmentation patterns are driven by ownership boundaries, roads, and other anthropogenic structures and patches of different ownerships are relatively homogeneous in forest structure and composition. Historical ecology is important from a landscape ecology perspective because despite changes to landscape structure in the last 150 years, the processes that operated on pre-settlement forest are still shared among the fragmented remaining forests (Ewers et al. 2013). Retaining the memory of processes that have successfully navigated changes in large-scale disturbance regimes in the past can help us identify important processes that need to be retained on the landscape if they are going to successful move forward.

In the second chapter, I found that both age-structure and ownership structure at edges was related to fire severity. Fire severity increased on public lands that were within 250 m of private land, most likely due to edge effects of salvage logging. More diverse age structure (larger difference in the age of dominant canopy trees) as fine scales (30 m) resulted in lower severity fire. Surface fuels alone are too variable at small scales to account for differences in fire behavior. The results of my study of fuel heterogeneity at two edge types with completely different structure and composition, found that fuels were heterogeneous within 50 m of both types of edges.

In the third chapter, I found that spotted owls use multiple fundamentally different fire-created edges at different spatial scales. At only very small scales (<0.8 ha), spotted owl used hard edges created by high severity fire adjacent to low severity fire. At broad scales (>12.9 ha), spotted owl selected for diffuse fire-created edges where the differences in fire severity were distinct, but subtler. Further testing of this finding with new data that combine post-fire use with field collected data on the structure of variable edges would help to define the parameters of used fire-created edges and may help develop management options for creating that beneficial edge structure.

Management implications

Taken together, these three studies suggest three common themes for land management.

Scale is important. The distribution of patch sizes at multiple scales is likely important to fire patterns, stand development patterns, and wildlife use patterns. At the smallest scale, individual structures, (i.e. large, old trees) that are fire resistant can serve as legacy components of planned disturbance. These legacies provide a cross-scale memory in the form of habitat components and seed sources. At stand-scales (<130 ha), fine-scale patchiness in age and canopy structure can promote fire resilience. It is probably not feasible to re-create through prescribed fire, wildfire, or management the frequency of disturbance that created the legacy components of the modern landscape. However, a concerted effort should be made to maintain the patchy structure of this forest type at landscape scales. At larger-scales (>500 ha), fire was less frequent, but still common.

Edges are not all created equally and they are important as drivers of disturbance and species distributions. In this dissertation, I quantified edges in several different ways and they showed unique relationships with different processes. The contour approach to quantifying changes in forest structure and fire severity underscores the importance of considering the biological function of an “edge” or “patch” before defining the structure of it. Different structures of edges have unique environments (e.g. competitive environments, resource availability, boundary layer protection). These unique properties can have fundamentally relationships with a process of interest.

Process and structure that worked in the past to reduce fire severity and support spotted owl populations are not static in time and space. Despite the depth of research that has quantified past fire patterns at multiple scales, in the huge realm of time and space, they are a drop in the bucket. The precise value of specific parameters of fire regimes are moving targets in time and space. Use the past to inform the present, but look to future needs and expected conditions to make management goals.

A landscape ecology approach to land management that includes concepts like landscape complexity and memory, patch size of disturbance, and landscape gradients could create balance between management objectives at landscape scales. For example, if we are interested in retaining forest types like oak woodlands and pine dominated mixed-conifer, we should consider the conditions under which they established in the past and try to emulate those conditions at appropriate scales and positions on the landscape. We, should use information about historic patterns to design management practices, but be flexible in the application of those practices over time. Trees are long-lived and species composition in any given age class generally reflects the conditions at the time of establishment. If management practices are not achieving measurable goals, they should be altered.

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APPENDIX: CHAPTER 2

Table A2-1. Difference in fine woody fuel loads between edge types (difference = abrupt- diffuse) for average difference across edges (difference = mature side- disturbed side) at each transect distance class.

Transect distance class	Mean difference between stand boundary types in average difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% confidence interval	upper limit of 95% confidence interval
5	0.10	0.29	0.36	23	0.7221	-0.68	0.89
10	0.12	0.29	0.41	23	0.6852	-0.66	0.90
25	0.14	0.29	0.50	23	0.6220	-0.64	0.93
50	-0.27	0.29	-0.94	23	0.3577	-1.05	0.51

Table A2-2Difference in fine fuel load from mature side to disturbed side for each transect distance class over both edge types.

Transect	Mean difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
5	-0.11	0.14	-0.78	11.5	0.4532	-0.54	0.31
10	0.18	0.14	1.25	11.5	0.2354	-0.25	0.61
25	0.11	0.14	0.79	11.5	0.4454	-0.31	0.54
50	-0.04	0.14	-0.28	11.5	0.7828	-0.47	0.39

Table A2-3. Difference in fine fuel load from the mature side to the disturbed side for each edge type.

stand boundary type	Mean difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
Abrupt	0.05	0.13	0.37	17	0.7133	-0.31	0.40
Diffuse	0.02	0.12	0.19	17	0.8494	-0.32	0.36

Table A2-4. Difference between abrupt and diffuse edges (difference = abrupt- diffuse) in average difference in live fuel load (difference = mature side- disturbed side) at each transect distance class.

Transect distance class	Mean difference between stand boundary types in average difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% confidence interval	upper limit of 95% confidence interval
5	-0.01	0.21	-0.06	23	0.9509	-0.58	0.56
10	-0.13	0.21	-0.62	23	0.5439	-0.70	0.44
25	-0.06	0.21	-0.28	23	0.7805	-0.63	0.51
50	-0.29	0.21	-1.39	23	0.1770	-0.86	0.28

Table A2-5. Difference in live fuel load from mature side to disturbed side for each transect distance class over both edge types.

Transect	Mean difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
5	-0.11	0.11	-1.08	11.5	0.3012	-0.43	0.20
10	-0.13	0.11	-1.28	11.5	0.2266	-0.45	0.18
25	-0.08	0.11	-0.81	11.5	0.4364	-0.40	0.23
50	-0.04	0.11	-0.38	11.5	0.7129	-0.35	0.27

Table A2-6. Difference in live fuel load for each edge type across all transects.

stand boundary type	Mean difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
Abrupt	-0.16	0.10	-1.50	17	0.1519	-0.44	0.13
Diffuse	-0.03	0.10	-0.32	17	0.7565	-0.31	0.25

Table A3-7. Difference between abrupt and diffuse edges (difference = abrupt- diffuse) in average difference in litter and duff fuel load (difference = mature side- disturbed side) at each transect distance class.

Transect	Mean difference between stand boundary types in average difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
5	-0.23	0.53	-0.43	23	0.6737	-1.68	1.22
10	-0.65	0.53	-1.21	23	0.2386	-2.09	0.80
25	-0.39	0.53	-0.73	23	0.4718	-1.84	1.06
50	-0.56	0.53	-1.04	23	0.3074	-2.01	0.89

Table A2-8. Difference in litter and duff fuel load from mature side to disturbed side for each transect distance class over both edge types.

Transect	Mean difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
5	-0.10	0.27	-0.37	11.5	0.7208	-0.89	0.69
10	0.39	0.27	1.44	11.5	0.1763	-0.40	1.17
25	-0.07	0.27	-0.28	11.5	0.7877	-0.86	0.72
50	0.58	0.27	2.17	11.5	0.0519	-0.21	1.37

Table A2-9. Difference in litter and duff fuel load for each edge type across all transects.

stand boundary type	Mean difference in fuel load	Std. Error	t value	DF	Pr(>|t|)	lower limit of 95% CI	upper limit of 95% CI
abrupt	-0.03	0.23	-0.13	17	0.8976	-0.66	0.61
diffuse	0.43	0.22	1.95	17	0.0680	-0.18	1.04

APPENDIX: CHAPTER 3

Defining diffuse and hard edges

We calculated slope of fire/logging severity (SLFS) using the slope calculator in ArcGIS (version 10). It calculated the horizontal and vertical rate of change in RdNBR for every pixel based on the change in value between the target cell and the eight surrounding cells (Figure A3.1), and the distance between cells (30m) to calculate a slope percent value. We then used fire/ logging severity thresholds (determined by MTBS) with three different threshold of SLFS (Tables A3.1, A3.2, and A3.3- see sensitivity analysis below) to identify hard (Figure A3.2) and diffuse edges (Figure A3.3). We then summed the number of hard (Figure A3.4) and diffuse edges (Figure A3.5) at six spatial scale from 0.8 ha to 829 ha.

Sensitivity analysis

In order to map the edges, I had to assign thresholds for “high”, “moderate”, and “low” RdNBR and SLFS. For fire severity thresholds, I used the thresholds for RdNBR set by MTBS. These thresholds are similar to values reported for other fires (the Biscuit fire and the Quartz fire) and are similar to thresholds that were validated for fires in the California Klamath mountains (Miller et al. 2009). Because fire severity is not our effect of interest, I did not test the sensitivity of the analysis to RdNBR thresholds for fire severity. We did run several iterations in order to test the sensitivity of the results to the specific threshold used for SLFS. Under the broadest definition (broad), I selected the threshold for moderate SLFS to be above the minimum SLFS that would represent no change across the boundary (100%). A change from the highest low-severity RdNBR (350) to the lowest high-severity RdNBR *600) would results in a 170% change (600/ 350*100= 170%). We used this for the threshold for high SLFS (Table A4.1). For a more narrow definition (Jenk’s definition), I used Jenk’s natural breaks of SLFS to identify three classes. Jenk’s natural breaks uses an algorithm to classify data into groups that minimize within group differences and maximize between group differences. This resulted in a moderate SLFS threshold of 209 and a high SLFS threshold of 534 (Table A3.2). Because diffuse edges were more common on the landscape, even under the narrower Jenk's classification, and I wanted to test a range of definitions, diffuse edges were identified at a third level that narrowed the range of SLFS further based on visual inspection (conservative diffuse). The moderate SLFS threshold for this narrow definition was 254.7 and the threshold for high SLFS was 435.5 (Table A3.3).

The results from the sensitivity analysis are in Table A3.4

Additional results

The results of the univariate model of spotted owl habitat selection as a function of habitat suitability only are listed in Table A3.5

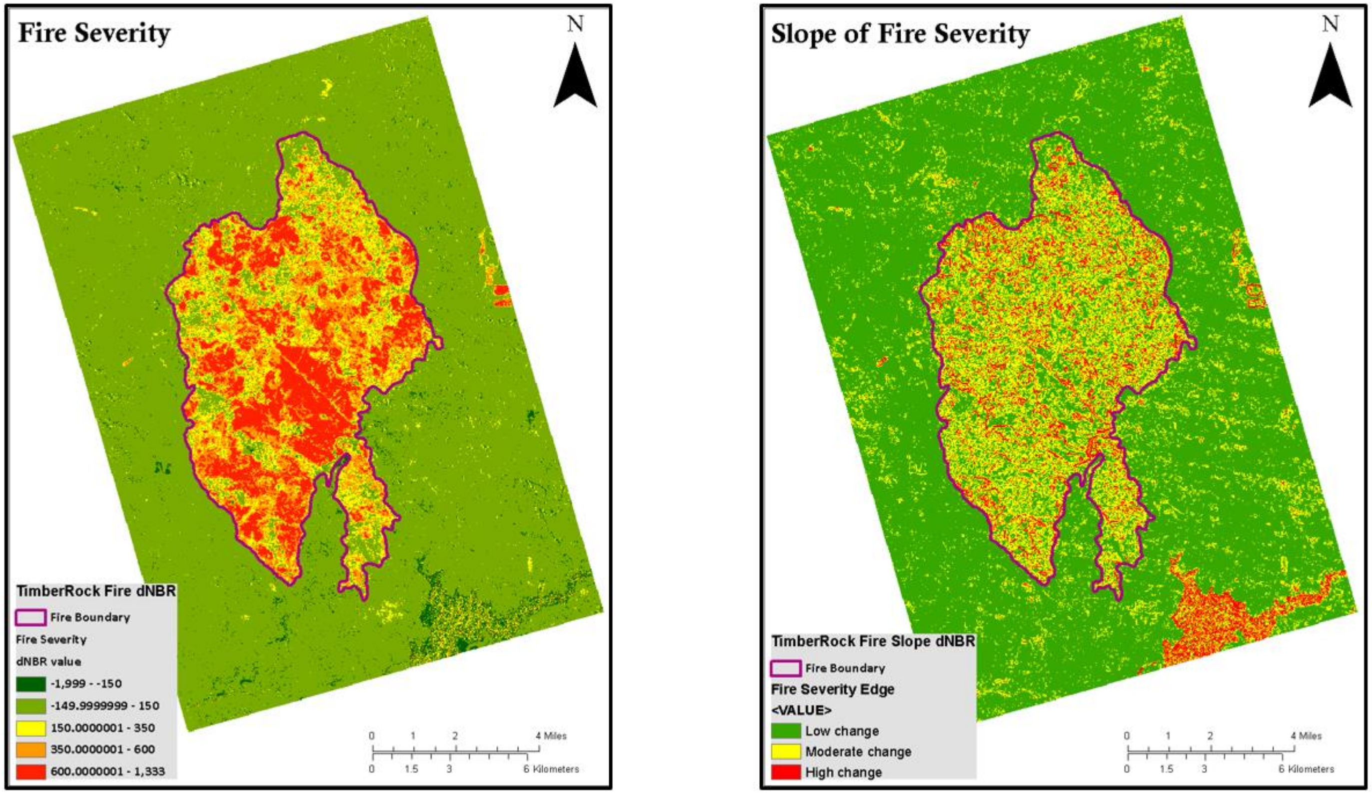

Figure A3.1 MTBS fire severity map (RdNBR) and calculated slope of fire severity map (SLFS).

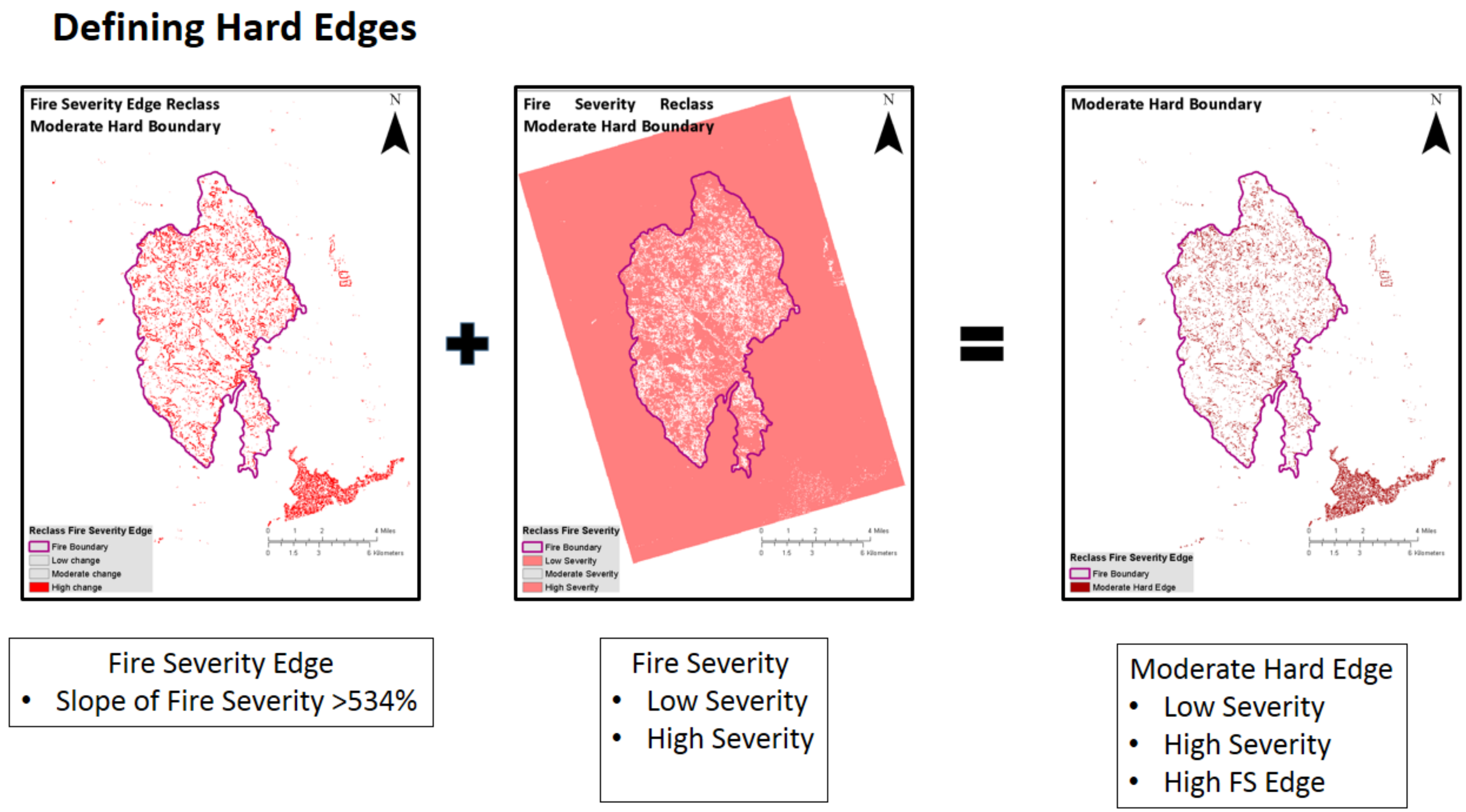

Figure A3.2. The steps used in defining hard edges under the moderate SLFS thresholds (Jenk's).

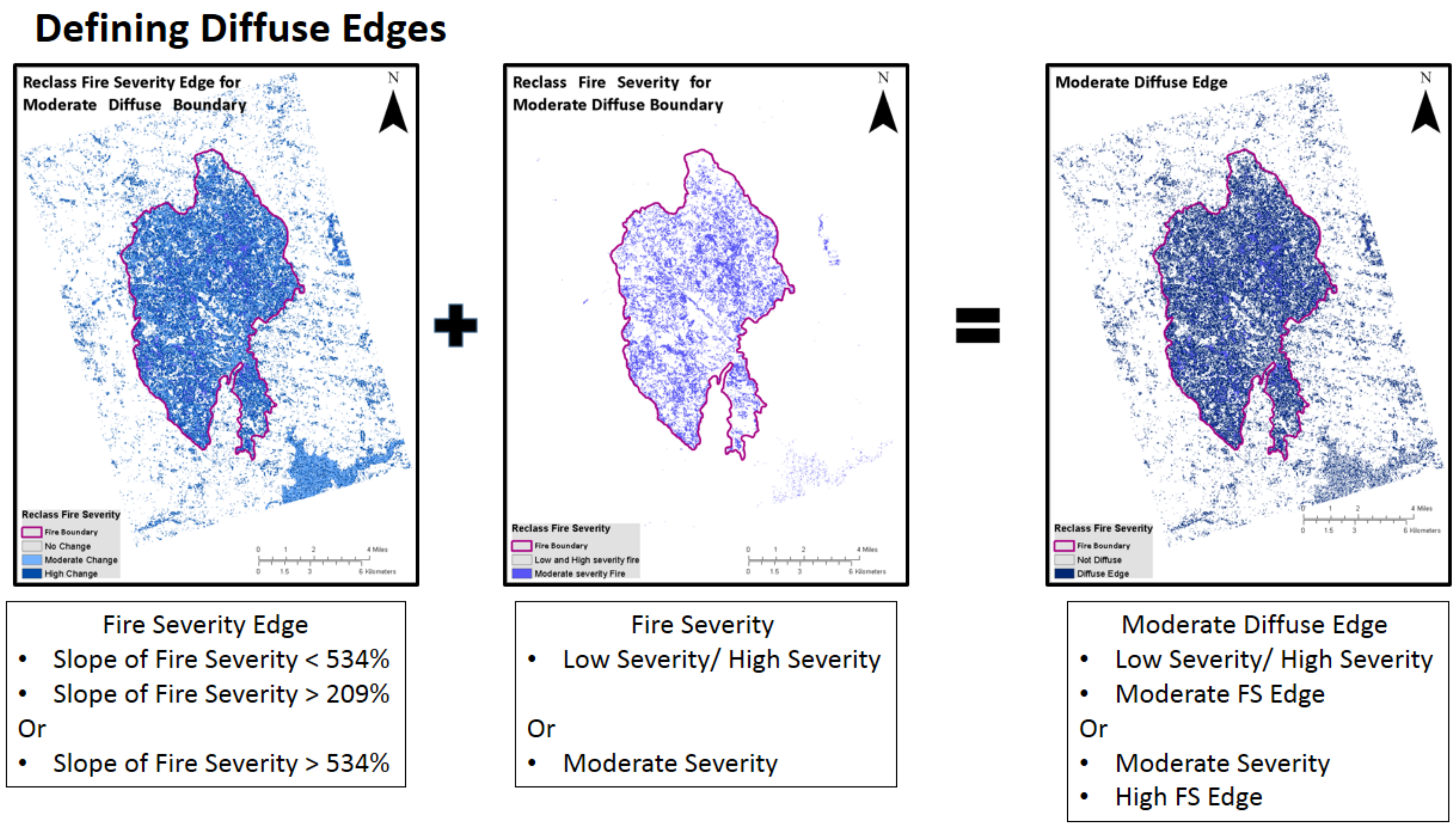

Figure A3.3. The steps used in defining diffuse edges under the moderate SLFS thresholds (Jenk's).

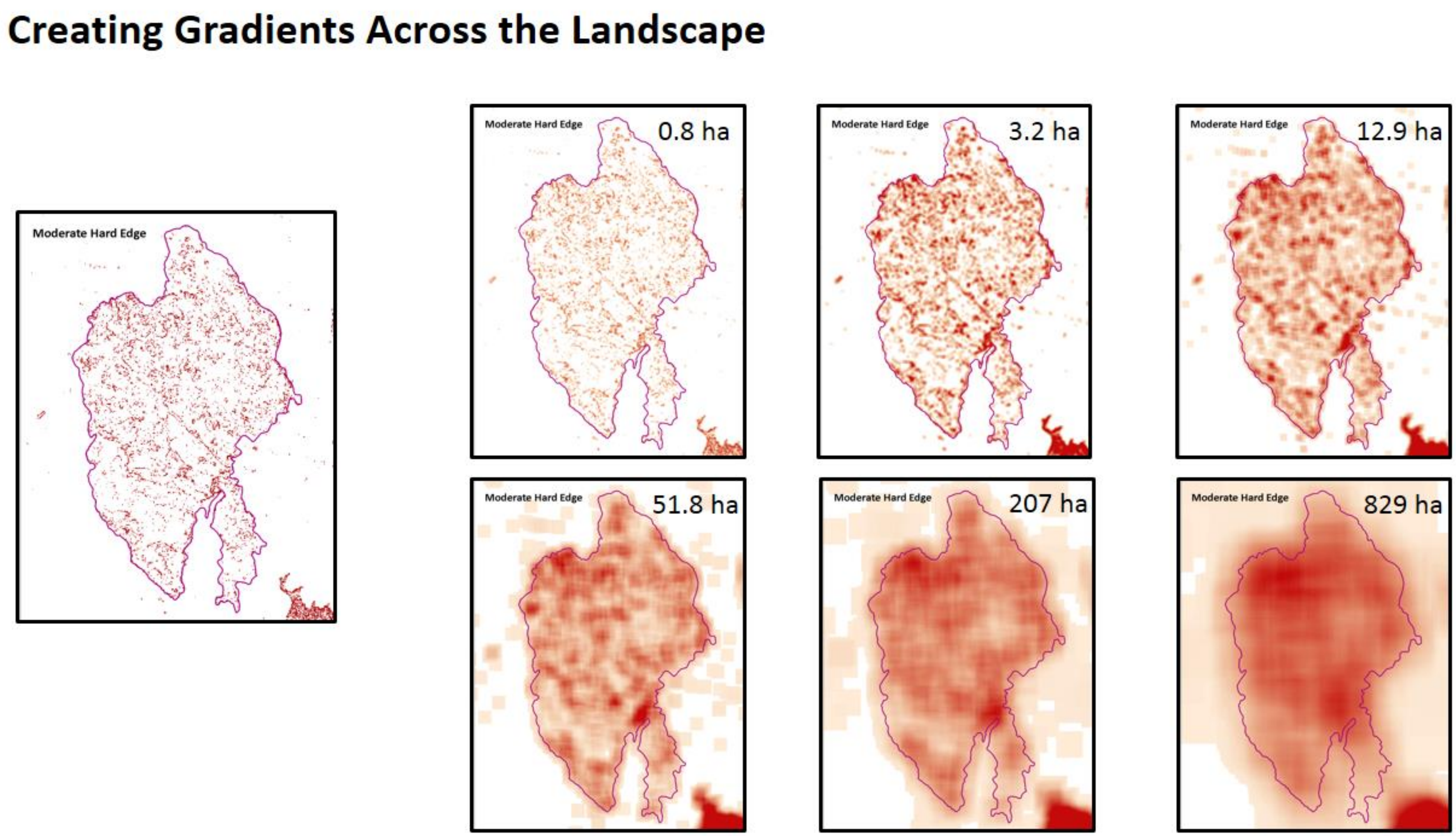

Figure A3.4. The number of hard edge cells at spatial scales varying from 0.8 ha (9X9 pixels) in the top left corner to 829 ha (96 X96 pixels) in the bottom right. Larger values are darker.

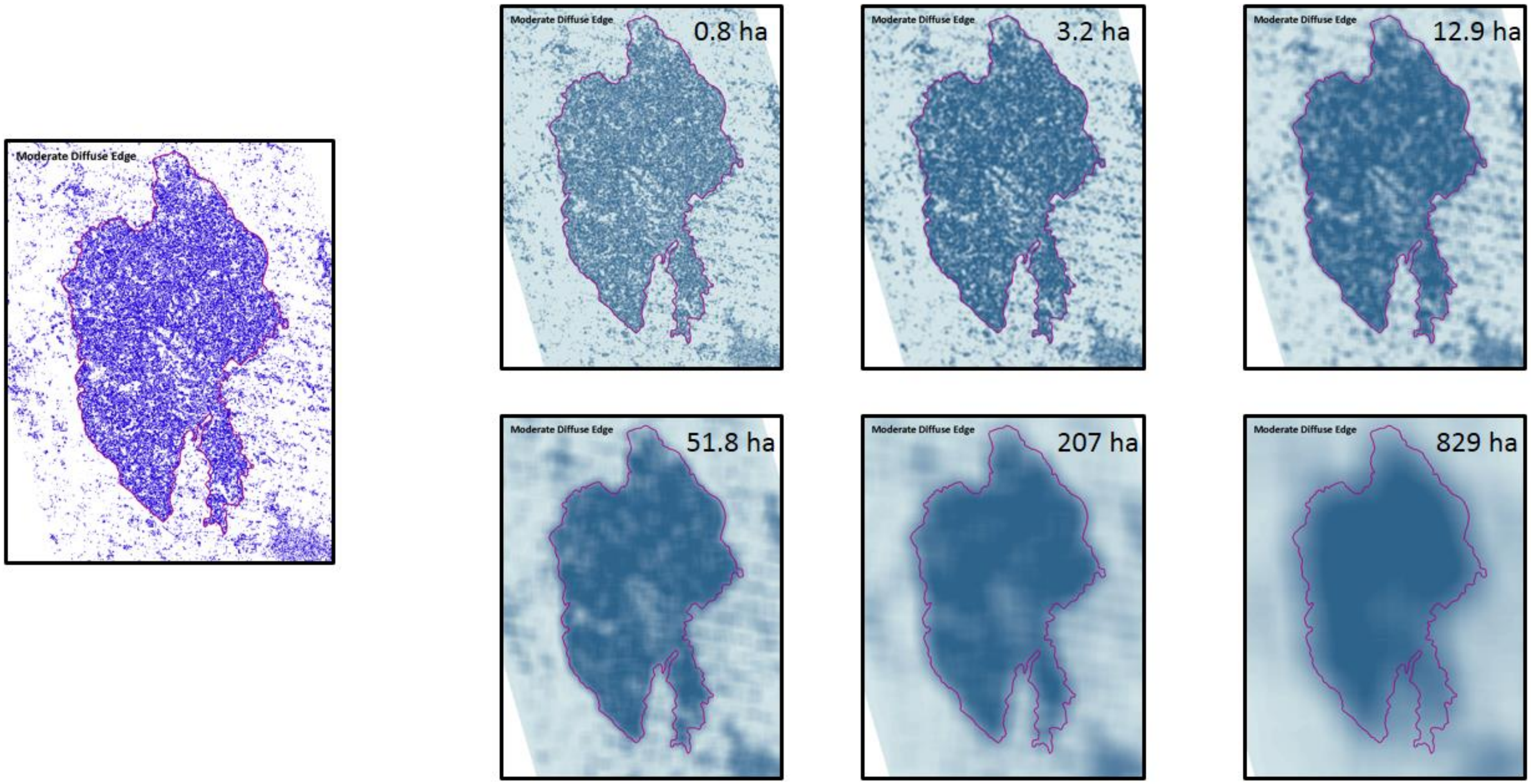

Figure A3.5. The number of diffuse edge cells at spatial scales varying from 0.8 ha (9X9 pixels) in the top left corner to 829 ha (96 X96 pixels) in the bottom right. Larger values are darker.

Table A3.1. Broad definitions of diffuse and hard edge were designed to catch all potential edges and were likely to include more edges than actually existed.

	Slope FS (%)		
FS	100	100-170	>170
less than -150	No Edge	Diffuse	Hard
- 150 - 150	No Edge	Diffuse	Hard
150-350	No Edge	Diffuse	Hard
350-600	No Edge	Diffuse	Diffuse
>600	No Edge	Diffuse	Hard

Table A3.2. Moderate definitions of hard and diffuse edges were based on Jenk's natural breaks to categorize the slope of fire severity.

	Slope FS (%)		
FS	209	209-534	>534
less than -150	No Edge	Diffuse	Hard
- 150 - 150	No Edge	Diffuse	Hard
150-350	No Edge	Diffuse	Hard
350-600	No Edge	Diffuse	Diffuse
>600	No Edge	Diffuse	Hard

Table A3.3. The conservative definition of diffuse boundary included a very narrow selection of slope of fire severity to restrict the definition.

	Slope FS (%)		
FS	<254.7	254.7-435.5	>435.5
less than -150	No Edge	Diffuse	
- 150 - 150	No Edge	Diffuse	
150-350	No Edge	Diffuse	
350-600	No Edge	Diffuse	Diffuse
>600	No Edge	Diffuse	

Table A3.4 Result of sensitivity analysis for different thresholds of hard and diffuse edge definitions. The results are of scaled models. The direction indicates whether it is a positive or negative relationship. It is starred if the variable is significant (p-value <0.05). The effect size is the β coefficient of a scaled model and indicates the relative magnitude of the effect. The model with the lowest BIC value is listed (competing models are listed below the top model), but the parameter values are only reported for the model, "probability of occurrence ~ Habitat Suitability Index (hab) + disturbance severity (FS) + amount of hard edge (h) + amount of diffuse edge (d).

		Habitat Suitability (hab)		Fire Severity (fs)		Hard Edge (h)		Diffuse Edge (d)		Best Model
Edge Definition	Scale (ha)	direction	effect size	direction	effect size	direction	effect size	direction	effect size	
Jenks	0.8	(+)*	0.89	(-)*	0.23	(-)	0	(+)	0.04	hab + FS
	3.2	(+)*	0.99	(-)*	0.36	(+)*	0.51	(+)	0.02	hab + FS + h +d
	12.9	(+)*	1.03	(-)*	-0.21	(-)*	0.28	(-)*	0.32	hab + FS + h +d
	51.8	(+)*	1.16	(-)*	0.16	(-)	0	(+)*	0.24	hab + fs*h +fs*d
	207	(+)*	1.24	(+)*	0.52	(-)*	0.88	(+)*	0.57	hab + fs*h +fs*d
	829	(+)*	1.1	(+)*	1.03	(-)	0.62	(+)*	0.15	hab + FS + h +d
Broad definitions of Hard and Diffuse Edge	0.8	(+)*	0.88	(-)*	0.21	(+)	0.05	(-)	0.02	hab + FS
	3.2	(+)*	0.94	(-)*	0.22	(+)*	0.1	(-)	0.01	hab + FS
	12.9	(+)*	1.06	(-)*	0.17	(+)*	0.08	(-)	0	hab + FS
	51.8	(+)*	1.17	(-)	0.01	(-)	0.01	(+)*	0.05	hab + FS
	207	(+)*	1.27	(+)*	0.6	(-)*	0.36	(+)*	0.02	hab
	829	(+)*	0.96	(+)*	0.82	(-)*	0.62	(+)*	0.4	hab + FS + h +d
Conservative Diffuse Edge	0.8	(+)*	0.89	(-)*	0.21	(+)	0	(+)	0.02	hab + FS
	3.2	(+)*	0.96	(-)*	0.49	(+)*	0.6	(+)*	0.87	hab + fs*h +fs*d
	12.9	(+)*	1.04	(-)	0.05	(-)*	0.22	(+)*	0.09	hab + fs*h +fs*d
										hab + FS + h +d
	51.8	(+)*	1.15	(-)	0.06	(+)	0.02	(+)*	0.17	hab + fs*h +fs*d
	207	(+)*	1.25	(+)*	0.81	(-)*	0.78	(+)*	0.2	hab + FS + h +d
										hab + fs*h +fs*d
	829	(+)*	1.1	(+)*	1.1	(-)*	0.58	(+)	0.04	hab + fs*h +fs*d

Table A3.5. Results of univariate model of spotted owl habitat selection as a function of average spotted owl habitat suitability (hab) within 0.8 ha, 3.2 ha, 12.9 ha, 52.8 ha.207 ha, and 829 ha of spotted owl and random locations.

	Habitat Suitability (hab)					
Scale (ha)	Sig	Probability	95% CI Lower Limit	95% CI Upper limit	BIC	ΔBIC
0.8	*	1.053	1.05	1.057	11326	205
3.2	*	1.058	1.054	1.062	11254	133
12.9	*	1.068	1.063	1.072	11121	0
51.8	*	1.079	1.074	1.084	11138	17
207	*	1.094	1.086	1.102	11551	430
829	*	1.073	1.063	1.083	12035	914