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Changes in forest structure and composition associated with unique land use histories: Implications for restoration

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CHANGES IN FOREST STRUCTURE AND COMPOSITION OF PONDEROSA PINE FORESTS OF THE NORTHERN ROCKIES WITH UNIQUE LAND USE HISTORIES: IMPLICATIONS FOR RESTORATION

by

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Abstract

Many contemporary semi-arid forests of western North America are denser and have a greater proportion of shade tolerant species relative to pre- Euro-American settlement. While many causes have been invoked to explain these changes, the active suppression of fire since the early 1900s has been the most widely studied and cited. However, widespread logging in western North American forests has often predated effective fire suppression and has affected a majority of semi-arid forests. The extent to which historical logging has contributed to uncharacteristically high densities and other changes in contemporary forests have never been adequately quantified. Therefore, true elucidation of the causes of departures of contemporary forests relative to historical conditions may be incomplete. I studied ponderosa pine/Douglas-fir forests of the Northern Rockies to address four main questions: 1) has historical logging exacerbated the effects of fire exclusion on forest density, structure and species composition?, 2) What is the magnitude of this change relative to that due to fire exclusion alone?, 3) in the absence of fire, which structural components in unlogged vs. historically logged stands are mostly responsible for deviations from reference ranges of variability?, and 4) what is the magnitude of such deviation in logged vs. unlogged forests? Based on a paired design (n=23 pairs) of logged, fire excluded stands with unlogged, fire excluded stands I found that fire excluded, logged stands were twice as dense as fire excluded, unlogged stands, and had higher numbers of small living and dead trees. While unlogged fire excluded forests generally experienced minimal to no departures relative to the range of stand densities observed in reference, fire-maintained stands, most logged fire excluded forests experienced substantial departures. Responses to the interaction of logging and fire exclusion varied by habitat type, with significant departures in Douglas-fir but not in ponderosa pine habitat types. The magnitude of the response was proportional to the intensity of historical logging. We suggest that unique restoration approaches are warranted for unlogged and logged, fire excluded forests and caution that fuel reduction and restoration policies which do not account for the legacy of logging may be ineffective in accomplishing their desired goals.

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INTRODUCTION

Semi-arid low and middle elevation forests of the western U.S. have experienced important changes since the settlement of Euro-Americans. These changes, including increases in tree density, shifts in species composition, and changes in resource availability, have occurred at both the landscape and stand level and are widely implicated as the cause of ecosystem dysfunction in many modern forests. Historically aberrant disturbances, diminished resiliency, altered structure and composition, and lack of spatial heterogeneity are all cited as evidence of this dysfunction and have become the focus of ecological restoration, perceived by many as the best solution for impaired ecosystem function. Because there is significant regional variation in the biological and abiotic controls of forest dynamics, successful restoration requires regionally specific information on the specific causes driving forest change, as well as their relative magnitude, to evaluate restoration needs and develop informed strategies to meet these needs.

Many forests have experienced multiple and often overlapping perturbations such as logging, grazing and the exclusion of fire, making it difficult to assess their relative effect on forest change. However, much less is known about the relative causal importance of these activities in forming contemporary forest conditions or the manner in which multiple management activities may interact to produce unique synergistic changes in forest characteristics and dynamics. Because few areas have been subject to only one type of historical management activity, this knowledge gap leaves us with a significant inability to identify the specific causes of forest change, which is fundamental to the accurate assessment of restoration needs. In this thesis I address part of this knowledge gap for ponderosa pine forests of the Northern Rockies bioregion. Specifically, my objectives are to: 1) evaluate the nature and magnitude of stand-level vegetation changes caused by fire exclusion alone relative to fire exclusion in combination with historical logging, and 2) use this information to develop a restoration strategy for forests with these distinct management histories. I address these questions in two chapters, described below.

Chapter 1: ***Strong effects of historical logging on contemporary structure of ponderosa pine forests of the Northern Rocky Mountains.***

This paper presents detailed stand structure and composition in logged and unlogged fire-excluded forests to compare the nature and relative magnitude of change experienced by sites with each management history. Although logging is generally acknowledged to have altered forest structure and composition by removing large, fire-tolerant trees, it is often asserted that fire exclusion has been responsible for the greatest changes in forest structure and composition. To test this notion, I use fire-maintained stands from a study by Keeling et al (2006) as a reference point of comparison to determine whether fire exclusion alone or fire exclusion in conjunction with historical logging has caused the greatest absolute changes in forest structure and composition from reference conditions. I then discuss the implications of our findings in the context of current fire and vegetation management policy and practice.

Chapter 2: ***Use of reference ranges of variability to evaluate restoration needs of unlogged, fire excluded and logged, fire excluded ponderosa pine/Douglas-fir forests of the Northern Rockies, USA.***

This paper uses the data of Keeling et al (2006) to explore ranges of variability of reference stand conditions and its application to the restoration of unlogged and logged fire-excluded ponderosa pine forests. The range of variability concept has become increasingly important in restoration efforts as a useful method of assessing when, and how, forest characteristics have experienced significant departures as a result of past management. I explore both the frequency and degree of departure from reference ranges of variability in logged and unlogged fire-excluded forests. Detailed comparisons of reference structure with that of logged and unlogged fire-excluded forests are then used to assess the specific components of stand structure and composition which have experienced departures and should therefore be the focus of restoration treatments. To evaluate the sources of variation in site density response to previous logging treatments, we examine the effect of site environmental variables and the intensity of past logging on site density response. Finally, we use this information to present a detailed set of prescriptive recommendations for restoration of unlogged and logged fire-excluded forests.

CHAPTER 1

STRONG EFFECTS OF HISTORICAL LOGGING ON CONTEMPORARY STRUCTURE OF PONDEROSA PINE FORESTS OF THE NORTHERN ROCKY MOUNTAINS.

Introduction

Many contemporary arid and semi-arid forests of western North America have been greatly altered since Euro-American settlement (Allen et al 2002, Hessburg et al 2005, Hessburg et al 2000, Kaufmann et al 2000, Keeling et al 2006, Covington 2000, Minnich et al 1995, Fule et al 2002, Arno et al 1995, Veblen & Lorenz 1986, Belsky & Blumenthal 1997, Baker et al 2007). These forests are frequently more homogeneous across spatial scales, are generally denser, but with fewer large trees and old growth stands, and have a greater proportion of ladder fuels and shade tolerant trees than historical forests (Allen et al 2002, Hessburg et al 2005, Hessburg et al 2000, Kaufmann et al 2000, Keeling et al 2006, Covington 2000, Minnich et al 1995, Fule et al 2002, Arno et al 1995, Veblen & Lorenz 1986,). While many causes have been invoked to explain these changes (Allen et al 2002, Hessburg et al 2005, Hessburg et al 2000, Kaufmann et al 2000, Keeling et al 2006, Covington 2000, Minnich et al 1995, Fule et al 2002, Arno et al 1995, Veblen & Lorenz 1986, Belsky & Blumenthal 1997, Baker et al 2007) the active suppression of fire since the early 1900s has been the most widely studied and cited (Allen et al 2002, Keeling et al 2006, Covington 2000, Minnich et al 1995, Fule et al 2002, Arno et al 1995). However, widespread logging in western North American forests has predated effective fire suppression by many decades and has affected a majority of arid and semi-arid forests (Minnich et al 1995, Fule et al 2002, Arno et al 1995, Veblen & Lorenz 1986, Baker et al 2007, Laudenslayer & Darr 1990, Brown et al 2004, Hessburg & Agee 2003). The extent to which historical logging has contributed to uncharacteristically high densities and other associated changes in many contemporary forests has never been adequately quantified. Therefore, true elucidation of the causes of such departures may be incomplete. Here, we quantify, for the first time, the interactive effects of historical logging and fire exclusion on contemporary forest structure and composition in ponderosa pine forests of the northern Rocky Mountains and compare these effects to those caused by fire-exclusion alone.

A clear distinction between the effects of fire suppression on current forest structure from the effects of additional factors such as grazing (Belsky & Blumenthal 1997) and logging is a necessary foundation for the development of restoration priorities, goals, prescriptions, and measures of success (Kauffman 2004). Because no attempt has been made to effectively differentiate the effects of fire suppression from the long-term effects of widespread historical logging the two factors have often been conflated. This has led to the common perception that increased forest density is primarily the result of decades of fire suppression alone, a notion which has come to serve as the basis for current forest management policies in the U.S. (White House 2002). However, existing circumstantial data suggest that logging may have contributed to increased stand density and abundance of shade tolerant species above those caused by fire exclusion alone (Kaufmann et al 2000, Minnich 1995). If so, the extent and magnitude of departures from historical reference conditions that have occurred during the fire exclusion period in logged and unlogged forests may differ. There is, therefore, a critical need to assess whether past logging has contributed to forest structural attributes commonly ascribed to fire exclusion alone and to quantify the relative magnitude of departures caused by fire exclusion effects with and without logging.

We used a paired design of logged, fire excluded stands (henceforth referred to as logged) with unlogged, fire excluded stands (henceforth referred to as unlogged) to quantify changes in forest structure and composition due to logging and fire exclusion while controlling for unrelated, confounding factors. We sampled a total of 46 stands (23 pairs) of low to mid elevation (avg.=1,296 m, range=946-1,753 m) pure and mixed ponderosa pine forests across Montana and Idaho. Sampled stands belong to low and mixed severity fire regimes but have not experienced fire for at least 65 years. Logged stands were harvested only once between 50-110 years ago by high grade logging, individual selection, or intermediate harvest methods. While no data were available on the extent to which understory trees were removed by logging treatments, extant stumps indicated that all logged sites experienced removal of some medium and large overstory trees (average % basal area of trees > 40 cm harvested = 62.17%, range = 15-100%).

Materials & Methods

Sample sites ranged from low to mid elevations and from dry sites of pure ponderosa pine stands to mesic sites where ponderosa pine is seral to many shade tolerant species. Of the 23 pairs of sites sampled, 2 were in the grand fir habitat type series, 17 were in the Douglas-fir series, and 4 were in the ponderosa pine series for MT (Pfister et al 1977) and ID (Steele 1981) forests. Fire regimes in ponderosa pine forests of the Northern Rockies include low and mixed severity regimes (Arno et al 1995, Heyerdahl et al 2008, Morgan et al 2008, Hessburg et al 2007). Fire scars were present on many trees in most of our sites, suggesting the historical presence of low severity fires. We did not conduct detailed fire history studies for our sites, but previous studies from within our study region confirm that both mixed and low severity fires occur in the forest types and regions that we sampled (9, 33-35).

A coarse analysis of potential watersheds for sampling was conducted using available GIS data layers of vegetation and disturbance (fire and logging) history and through direct identification by Forest Service silviculturists and fire specialists. Fire history layers extended back to 1940. Using this year as a minimum threshold for time since fire, none of our sites had experienced fire for at least 65 years although many more decades may have passed for most sites since the last fire. Logging history layers generally dated back no further than the 1950s. Information on historical logging predating the 1950s was collected from local Forest Service staff and was used to supplement the GIS layer to identify watersheds which historically experienced timber harvest. Selection criteria to identify sites for potential paired plots included: no known grazing history, lack of fire for at least 60 years, a single logging event no more recent than 50 years old, and close proximity and similarity of physiographic parameters between paired stands.

Extensive field surveys were conducted in the initially selected watersheds to identify and select specific suitable paired stands. All sites were surveyed for signs of recent grazing or fire, for the presence of old stumps in logged sites and the absence of stumps or other signs of previous harvest in unlogged stands, and for the presence of suitable pairs within the same historical stand or in neighbouring stand with similar physiographic characteristics. Given the general lack of detailed historical grazing records, we cannot be certain that grazing never occurred on any of our sites. However, it is highly unlikely that there was any systematic bias towards grazing in logged or unlogged sites, given the paired sampling design we employed. To ensure that paired stands belonged to the same historical stands, attention was paid to finding sites where pre-harvest tree size and age in logged stands (estimated from stumps and remnant trees), was similar to large tree size and age in unlogged stands. Age was determined using tree cores. In logged sites, the relative decay of stumps was visually assessed to determine whether multiple entries had been made into a stand. If stumps of similar species and size classes were found in distinct and separate phases of decay then we determined that multiple logging events had occurred and the stand was deemed unsuitable for sampling.

One 20 m x 50 m plot was randomly placed within each stand with its long axis perpendicular to the slope. Physiographic site variables including slope, aspect, elevation, and habitat type were recorded at plot center. Within each 20 m x 50 m plot, the diameter at breast height (DBH) of all trees (≥ 4 cm DBH), the number of seedlings (< 4 cm DBH and < than 5 m tall) and saplings (< 4 cm DBH and ≥ 5 m tall) and the DBH of dead trees was measured and recorded by species. When present, other tree species such as lodgepole pine, grand fir, and limber pine, were included in pooled species statistics, but were not calculated separately due to their absence from many stands and low abundance within stands where present. Stump diameters were measured at the highest feasible point on the stump and the height of measurement was recorded. Sections were removed using a chainsaw from the most viable portion of each stump. However, significant outer sapwood decay on stumps prevented the use of dendrochronological crossdating methods to determine specific logging date. Therefore, when available, we consulted with local Forest Service historians, silviculturists and fire specialists to estimate the approximate harvest date of logged stands. Harvest dates for our sites ranged from the mid 1890s- mid 1950s.

<u>Statistics</u>

For all normally distributed data, paired t-tests (n=23) were used to evaluate statistical differences between logged and unlogged stands for plot elevation, total density, total BA, and the size class distributions of density and BA for each species individually, for snags and for all species pooled. When necessary, ln transformations were used to meet normality requirements. Non parametric Wilcoxon signed rank tests were used to assess differences in BA and density in three cases where data was not normally distributed and could not be transformed to meet normality requirements. Bonferroni corrections (0.05/5 = .01) were used to account for multiple t-test comparisons of density and BA between the five size classes within each group (ponderosa pine, Douglas-fir, all species pooled, and snags). Numeric results are included in Supplementary Information Table 1, with p-values listed in Supplementary Information Table 2.

Independent sample t-tests based on equal or unequal variances as necessary were used to compare total density and density by species between unlogged, fire-maintained stands from Keeling et al. (2006) (n=6), unlogged, fire excluded stands (pooled values from Keeling et al. 2006 and this study) (n=29), and logged, fire excluded stands from this study (n=23). Data for species other than ponderosa pine and Douglas-fir were not normal and Mann-Whitney non parametric comparisons were used to test treatment differences. Treatment comparisons are conservative because they do not account for pairing of stands. α values were set at 0.05.

Results

Average total stand density of logged stands was more than twice that of unlogged stands ($p < 0.001$, Fig. 1a), although there was significant variation in the density of both logged and unlogged stands (Supplementary Information Table 1). Higher density in logged stands was due to a significant increase of small (DBH < 40 cm) trees ($p < 0.001$, Fig. 1a). In contrast, the density of large (DBH > 60 cm) trees was significantly lower in logged stands relative to unlogged stands ($p = 0.001$, Fig. 1a). The density of ponderosa pine trees < 40 cm DBH was higher in logged stands than in unlogged stands, although differences were statistically significant only for trees 4-20 cm DBH ($p < 0.01$, Fig. 1b). As a result there was a more even distribution of ponderosa pine tree density across all size classes in unlogged stands (Fig. 1b). The total density of Douglas-fir was higher in logged stands, with significant differences for trees 20-40 cm ($p < 0.01$, Fig. 1c). However, in both logged and unlogged stands the proportion of total

Douglas-fir density comprised of trees < 40 cm DBH was similar (Supplementary Information Table 1), likely a result of their shared history of fire exclusion. The total density of snags was significantly higher in logged stands than in unlogged stands ($p < 0.01$), due mainly to large numbers of small snags in logged stands ($p < 0.01$, Fig. 1d).

There were no statistically significant differences in total stand basal area between unlogged and logged stands ($p > 0.1$, Fig. 2a). However, basal area of ponderosa pine tended to be lower in logged stands while that of Douglas-fir was higher (Supplementary Information Table 1). Furthermore, basal area distribution among size classes was substantially different between logged and unlogged stands. Basal area in the largest size class (> 60 cm DBH) was significantly higher in unlogged stands ($p < 0.001$) due to the abundance of large ponderosa pine trees (Fig 2b). In contrast, basal area in smaller size classes (4-20 cm and 20-40 cm DBH) was significantly higher in logged stands ($p < 0.001$, Fig. 2a) due to higher BA of small Douglas-fir ($p < 0.01$ for 4-20 cm class) and ponderosa pine trees ($p < 0.01$ for 20-40 cm class) (Fig. 2b, c).

Our results show that relative to unlogged stands, logged stands were denser, with greater numbers of smaller living trees and small standing dead trees, and fewer large trees. As a result, logged stands exhibited less structural diversity than unlogged stands. An important question, however, is whether the logging effect in stands not subjected to fire is quantitatively significant relative to the effects due to the absence of fire alone. This comparison is fundamental to an understanding of the factors driving current forest structural departures from historical conditions. If the combined effects of logging and fire exclusion are large relative to the effects of fire exclusion alone, then the common claim that fire exclusion alone is primarily responsible for changes in forest structure in ponderosa pine forests of the Northern Rockies may not be warranted.

The paucity of fire-maintained stands outside wilderness or remote areas precluded comparison of a full complement of paired frequently burned and unburned stands both with and without logging. Instead, we used the data of Keeling et al (2006) who quantified fire exclusion effects in ponderosa pine forests of the northern Rocky Mountains (average elevation of 1,233 m) by pairing unlogged stands subjected to two to four fires in the 20th century ("burned") with unlogged stands not burned for at least 74 years. The sample stands from Keeling et al (2006) encompassed a smaller geographic area than our study which included some dry, pure ponderosa pine stands. Therefore, while not ideal, our coupled data sets provide a unique comparison of stand attributes across unlogged, fire-maintained stands and logged and unlogged, fire excluded stands. Total density and density of ponderosa pine and Douglas-fir between unlogged stands from Keeling et al (2006) and our study were not statistically different ($p > 0.05$ for all variables, Fig. 3) although slightly higher abundance of other shade tolerant species in Keeling et al (2006) resulted in slightly higher, but not statistically different total density. To further test the validity of pooling unlogged stands from both studies, we conducted a subset analysis of the eight pairs of sites from within our study region that also fell within the sub-region studied by Keeling et al (2006). No statistically significant differences ($p < 0.05$ for all variables) were found between this subset of our unlogged sites and those of Keeling et al (2006) for any tree species groups. Therefore, it is unlikely that differences reported here between the full set of our sites and those of Keeling et al (2006) are due simply to geographic variation of stand characteristics across our broad study region instead of effects associated with their unique land use histories. Comparisons between burned, unlogged, and logged stands were similar to those reported above when pooling sites from Keeling et al (2006) with either our full data set or the subset data. The consistency between these results is evidence that the juxtaposition of the two data sets is valid.

Examination of the combined data sets allows the effects of fire-exclusion alone to be distinguished from the interactive effects of fire-exclusion with historical logging (Fig. 3). Relative to burned stands, total stand density increased 1.7 fold due to fire-exclusion alone ($p = 0.05$; 5), a result of the increased abundance of Douglas-fir ($p = 0.03$) and a tendency towards higher density of other shade tolerant species. Total density increased 3.3 fold due to the combined effects of fire-exclusion with historical logging ($p < 0.001$), which resulted from significant increases of Douglas-fir ($p < 0.01$) and a tendency for higher density of all other species.

Discussion and Conclusions

We show that historically logged, fire excluded ponderosa pine forests of the Northern Rocky Mountains, USA have much greater average stand density, standing dead trees and abundance of fire intolerant trees than unlogged, fire excluded counterparts. Further, the interactive effects of logging and fire exclusion far exceed those due to fire exclusion alone. While fire exclusion in many arid and semi-arid forests of the western U.S. has certainly led to increased average forest density, primarily due to increases in small shade tolerant species, the rate and magnitude of this change is highly variable (Keeling et al 2006). Timber harvesting in forests of the northern Rocky Mountains, however, has greatly exacerbated increases of small tree density, to the extent that logged forests bear little resemblance either to modern, unlogged fire-excluded forests or their historical, fire-maintained counterparts. Given the extensive history of logging in arid and semi-arid forests across the western U.S. (Minnich et al 1995, Fule et al 2002, Arno et al 1995, Veblen & Lorenz 1986, Baker et al 2007, Laudenslayer & Darr 1990, Brown et al 2004, Hessburg & Agee 2003), this finding represents a previously under recognized but critical contribution to our understanding of the factors driving current departures of forest conditions from those which existed prior to Euro-American settlement. Similar to substantial regional variation in fire regimes and fire exclusion effects in arid and semi-arid western forests, we note that the effects of logging may also vary across broad geographic regions. For example, limited data from southwestern ponderosa pine forests suggests that logging may not produce the long-term density effects we document in the Northern Rockies (Fule et al 2002). Further research is needed to assess regional differences in the effects of logging relative to other anthropogenic disturbances responsible for current departures of forest condition from historic ranges of variability.

At least for ponderosa pine forests of the Northern Rockies, a region where recent fire activity has expanded greatly (Westerling et al 2006), our results challenge the common assumption that fire exclusion alone has been the primary factor responsible for the dramatic increases in stand density, ladder fuels, and the abundance of shade tolerant tree species typical of many contemporary forests. This, combined with the accumulation of dead trees and residual logging slash which serve as surface fuel (Dodge 1972, Agee 1993, Skinner & Chang 1996), suggests that previously logged stands are likely more prone to severe, stand replacing wildfire and insect disturbances than either unlogged, fire excluded or fire-maintained stands (Weatherspoon & Skinner 1995, Odion et al 2004).

This is consistent with reports of uncharacteristically severe fires in contemporary, previously logged forests (Baker et al 2007, Dodge 1972, Agee 1993, Skinner & Chang 1996, Weatherspoon & Skinner 1995, Odion et al 2004, Steele et al 1986) in the Northern Rockies and elsewhere in the western U.S. These findings highlight a significant ecological and social cost resulting from commercial timber extraction that has been poorly recognized and frequently, but incorrectly, ascribed to fire suppression.

There are notable implications of our results for restoration of arid and semi-arid forests of the Northern Rockies. First, they provide convincing evidence that previously logged ponderosa pine forests of the Northern Rocky Mountains have experienced greater departures from historical conditions than unlogged, fire excluded forests. While previously logged, fire excluded forests may, therefore, require significant mechanical stand manipulations before fire can be safely introduced, unlogged, fire excluded forests may require much less invasive treatments. In fact, growing evidence suggests that labor intensive and costly mechanical vegetation manipulation in previously unlogged forests prior to the restoration of natural wildfires may be unnecessary, and potentially counterproductive, despite structural departures from historical conditions (Collins et al 2007a, Collins et al 2007b, Odion & Hanson 2006, Fule & Laughlin 2007).

Second, our results point to potential long-term risks associated with mechanical treatments of stand structure, especially in previously unlogged forests. While modern silvicultural techniques designed for fuel reduction may not mirror historical logging practices, both involve soil disturbance and changes of canopy cover. The extent to which mechanical fuel reduction may have similar long-term negative effects to those reported here when treated stands are left unattended is unknown. Such lack of scientific evidence incorporates a fundamental element of risk, particularly if *recurrent* fire is not restored to treated stands. The successful reintroduction of fire is contingent on the long-term commitment of financial resources and consistent management policy that promotes a greater use of prescribed and wildland fire on a *landscape* scale. Currently, where over half of the Forest Service budget is spent on fire suppression and other wildfire-related activities and 97-99% of all fires are purposefully extinguished (Kauffman 2004, Stephens & Ruth 2005), it is clear that neither the financial resources nor the policy imperatives for such a commitment have yet been put in place. In light of our results and in the absence of this commitment, a cautious approach to landscape level mechanical fuel reduction treatments in previously unlogged stands as a form of ecological restoration is warranted. In contrast, emphasizing the greater need for fuel reduction treatments in second growth forests, especially near communities and existing road infrastructure where treatment maintenance and monitoring is most feasible, will likely maximize the efficiency and economy of restoration treatments and minimize ecological risks.

CHAPTER 2

USE OF REFERENCE RANGES OF VARIABILITY TO EVALUATE RESTORATION NEEDS OF UNLOGGED, FIRE EXCLUDED AND LOGGED, FIRE EXCLUDED PONDEROSA PINE/DOUGLAS-FIR FORESTS OF THE NORTHERN ROCKIES, USA.

Introduction

Widespread changes in forest conditions as a result of the last century of land use have sparked significant interest in the potential for ecologically based restoration of western U.S. forests (Brown et al 2004, Allen et al 2002, Noss et al 2006, Covington 2000, Covington & Moore 1994, Hessburg et al 2000). Land uses including grazing (Bakker et al 2007, Belsky & Blumenthal), logging (Naficy Chapter 1, Fule et al 1997, Veblen & Lorenz 1986, Baker et al 2007, Kauffman et al 2000, Laudenslayer & Darr 1990, Minnich et al 1995, Hessburg & Agee 2003), and fire exclusion (Covington & Moore 1994, Fule et al 2002, Baker et al 2007, Keeling et al 2006, Minnich et al 1995, Kauffman et al 2000, Brown et al 1999) are thought to be among the primary causes of the changes in forest structure and composition since the arrival of Euro-Americans to the western U.S. It has been suggested that recent increases in the aerial extent and severity of natural disturbances such as wildfires and insect epidemics are a consequence of structural and compositional changes induced by Euro-American management (Allen et al 2002, Covington 2000, Fule et al 2004, Hessburg et al 2005, Goforth & Minnich 2008, Hessburg & Agee 2003). Restoration of forest structure and composition to some approximation of the conditions which predated Euro-American settlement has been postulated as an important part of minimizing aberrant disturbance patterns (Agee & Skinner 2005, Brown et al 2004, Allen et al 2002, Moore et al 1999, Covington 2000). Increasingly, the impaired conditions of many forest ecosystems as a result of past management have also become an important basis for national forest policies which promote further active management in general and forest restoration in particular.

Forest restoration research and practice have tended to focus on ameliorating departures in stand structural and compositional attributes from historical reference conditions. Most commonly, reference conditions in forest systems are determined from stand reconstructions of contemporary forests to the pre-settlement era (Covington & Moore 1994, Mast et al, Arno et al 1995). Late 19th and early 20th century forest survey records and studies have also been utilized in the fortunate cases where they exist (Moore, Baker et al 2007, Minnich et al 1995, Stephens 2000). Relatively unimpaired contemporary forest systems are rare in the landscape, but in few instances they have been identified and used to establish reference conditions (Keeling et al 2006, Minnich et al 2000, Stephens & Fule 2005, Stephens & Gill 2005, Fule et al 2002, Fule & Covington 1998). In all cases, reference conditions, or the range of observed conditions, are applied to three critical questions: 1) when has a forest system been significantly changed by management activities (i.e. is in need of restoration), 2) what changes have occurred and 3) what target range of conditions should be aspired to as restoration goals for the impaired forest landscape.

An important recognition that has emerged from the body of conservation biology and restoration ecology is that departures from reference conditions cannot accurately be measured from average metrics of system characteristics, but must incorporate a meaningful measure of the range of variability of reference conditions (White & Walker 1997, Landres et al 1999, Swetnam et al 1999, Hollings & Meffe 1996). The utility of reference ranges of variability lies in its contribution to the inclusion of natural rates of change and dynamic states that characterize all natural systems, but especially disturbance mediated systems, rather than adherence to arbitrary time frames or static conditions at a point in time (Hollings & Meffe 1996, Swetnam 1999). Additionally, the range of variability concept is valuable because in capturing the range of variation observed in reference systems, it is potentially diagnostic and insightful of the processes which foster such variability and are thus fundamental to ecosystem function (Odion & Hanson 2006, Hollings & Meffe 1996, SER Primer). The concepts of reference, natural or historical ranges of variability have therefore become an increasingly important framework in ecological restoration theory and will derive great benefit from detailed quantitative data that describes and allows comparison of patterns of variability associated with reference sites and sites in need of restoration.

In addition to accurate knowledge of the range of variability in reference sites the range and specific nature of departure associated with distinct land management activities is fundamental to the evaluation of restoration needs (Noss et al 2006, Kauffman 2004). Not all land uses have equivalent effects (Naficy Chapter 1, Baker et al 2007), and they may interact in non-additive ways to produce long-lasting, novel effects or ecological surprises (Paine et al 1998, Bakker et al 2007, Naficy Chapter1). Given the broad variety of land use histories, unique, case-specific restoration prescriptions or approaches are likely warranted. Information on the range and nature of departures associated with distinct management histories will also greatly benefit broad scale assessment of restoration priorities.

Low and mid elevation, semi-arid forest landscapes of the western U.S. were historically characterized by high frequency disturbance regimes and exhibited high levels of structural variability (Stephens & Gill 2005, Cooper 1960, West 1969, White 1985). Due to their accessibility, Euro-American settlement and associated land management activities have likely caused the greatest disruptions of natural disturbance processes and changes in community composition, structure, and function in these systems. Consequently, these low and middle elevation forest systems have become a major focus of restoration research and practice. Among these forest types, ponderosa pine (*Pinus ponderosa*) forest covers a vast geographic area extending across a range of elevations from southern Canada to northern Mexico, and has therefore been one of the most intensively studied forest systems. A large body of work has investigated historical and contemporary conditions for ponderosa pine forests of the southwest (Cooper 1960, White 1985, Covington & Moore 1994, Fule 1997), the Black Hills (Shinneman & Baker 1997, Brown & Cook 2006, Brown & Sieg 1996), the Central Rockies of Colorado's front range (Brown et al 1999, Kaufmann et al 2000, Veblen & Lorenz 1986, Veblen et al 2000, Sherriff & Veblen 2006), the eastern Cascades (Wright & Agee 2004, West 1969), California's southern and eastern mountain ranges (Savage 1994, Stephens 2000, Minnich et al 1995), and a few areas in northern Mexico (Stephens & Fule 2005) and Canada (Heyerdahl et al 2007). These and other studies have been used to establish reference stand conditions and ranges of variability for ponderosa pine forests that serve as the basis for assessments of restoration needs and appropriate goals.

In contrast, relatively few studies exist of reference ranges of variability in the widespread ponderosa pine forests of the Northern Rockies (Arno et al 1995, Keeling et al 2006). This lack of regionally-specific reference data is significant and problematic, as substantial regional variation in ponderosa pine forest dynamics have been documented in association with unique regional climates, disturbance regimes, successional processes, and competitive environments (McKenzie et al 2000, Schoennagel et al 2004). Because historical ecological data are widely used to establish reference conditions and ranges of historical variability, understanding these regional differences in ponderosa pine forest dynamics is fundamental to the effective assessment of restoration priorities, appropriate prescriptions, target goals, and measures of success (Swetnam et al 1999).

In ponderosa pine forests of the Northern Rockies, we have shown that fire exclusion interacts strongly with logging disturbance, producing substantially different stand conditions in logged versus unlogged stands (Naficy Chapter 1). While the results of Naficy (Chapter 1) corroborate the findings of other studies showing important variation in stand level effects of fire exclusion, they found that logging in combination with fire exclusion not only produced a higher average departure from historical reference conditions, but exhibited much greater variability as well. While some studies have evaluated the potential sources of variation in stand structure due to the lack-of-fire (Keeling et al 2006), we are aware of no studies that have attempted to do so in logged, fire excluded forests. Here, we present stand structural data from unlogged fire-maintained stands, logged fire excluded and unlogged fire excluded ponderosa pine forests of the Northern Rockies. The objectives of our research were: 1) to provide a detailed description of stand structure and composition of contemporary unlogged, fire-maintained sites to be used as reference sites, with explicit attention to the observed ranges of variation in stand conditions, 2) to evaluate the frequency and degree of departure from reference ranges of variation in unlogged, fire excluded and logged, fire excluded forests, 3) to identify specific components of stand structure and composition that have departed as a result of fire exclusion alone and in combination with logging and should therefore be targeted in restoration treatments, 4) to assess the factors which contribute to the observed variation in degree and frequency of departures from reference ranges of variation in logged, fire excluded sites and compare this with known influences on variability of unlogged, fire excluded sites 5) to outline an approach to restoration of unlogged and logged, fire excluded ponderosa pine forests of the Northern Rockies that accounts for variability in reference ranges of variation and the factors which influence the frequency or degree of these departures.

Methods

Our study area included portions of central and western Montana as well as central Idaho. Data presented here were pooled from two complementary studies within this region. Keeling et al (2006) used a paired study design to compare forest characteristics of contemporary unlogged, fire-maintained (n=6) and paired unlogged, fire excluded (n=6) ponderosa pine forests in central Idaho and western Montana. Fire-maintained sites in Keeling et al (2006) were defined as sites that had experienced between 2-4 fires since 1880, whereas their fire excluded sites had not experienced any fires during the same time period. Naficy (Chapter 1) used a similar design comparing unlogged, fire excluded forests (n=23) like those studied by Keeling et al (2006) paired with logged, fire excluded forests (n=23) across a broader portion of Montana and Idaho. In both studies, pairing of stands was done so as to minimize differences in physiographic conditions (elevation, aspect and slope). At each stand structural information was collected by measuring the diameter at breast height (DBH) of all live and dead trees found within 20m x 50m plot. All stumps within logged sites were also measured for DBH and recorded by species. Density, species composition and size class information was then derived from these data. See Keeling et al (2006) and Naficy (Chapter 1) for greater detail on experimental design and field methods.

Using these data, we calculated two measures of average conditions (mean and median), several metrics of variability (interquartile range, variance, standard deviation, and range), and created size class distributions for stand density of unlogged, fire-maintained (“burned”) sites, for unlogged, fire excluded (“unburned”) sites, and for logged, fire excluded (“logged”) sites. In order to preserve information resulting from the paired design of each study, we kept unburned sites from Keeling et al (2006) and Naficy (Chapter 1) separate. All calculations were made for total stand density and also for density data partitioned by size class and by major tree species groups. We organized data into four size classes: 5-20 cm, 20-40 cm, 40-60 cm, and > 60 cm and four tree species groups: ponderosa pine, Douglas-fir (*Pseudotsuga menziesii*), other species, and all species pooled. Species other than Douglas-fir and ponderosa pine included grand fir (*Abies grandis*), lodgepole pine (*Pinus contorta*), subalpine fir (A*bies lasiocarpa*), and Rocky Mountain Juniper (*Juniperus communus*).

Ranges of variability and departure calculations

We used the interquartile range (IQR), defined as the 75th minus the 25th percentile (i.e. the middle 50% of density values), and the full range (maximum-minimum) of density values to represent a conservative and broad measure, respectively, of the range of variability in stand density associated with each treatment. To assess the frequency of departure from reference conditions (burned sites), the percentage of sites within each treatment that fell outside the reference range of variability in density parameters for each species group was calculated. We also tallied whether departed sites fell above or below reference ranges of variability. Departures above the reference ranges of variability (i.e. where unburned or logged sites had higher density than reference ranges of variability) were termed "positive departures", while departures below the reference ranges of variability were termed "negative departures". For those sites which did depart from reference ranges of variability, the degree of departure was compared relative to the outer limit of the reference interquartile range or the full range. The median, mean and range of the degree of departure were then tabulated for all stands in each treatment that fell outside the reference range.

Stand structure & composition between burned, unburned and logged sites

Density differences by size classes and species groups between unburned sites from Keeling et al (2006) and those in Naficy (Chapter 1) were first tested with independent samples t-tests to determine whether unburned sites from both studies could be pooled into a single unburned treatment. No differences between unburned sites from either study were found ($p > 0.1$ for all comparisons). We therefore pooled data for unburned sites from both studies in all subsequent statistical comparisons with burned and logged sites. Statistical differences between treatments with respect to all size class and tree species groups were tested using independent samples t-tests for all normally distributed variables. Square root and natural log transformations were used where necessary to meet normality requirements. For variables that could not be transformed to meet normality assumptions of parametric tests, non-parametric Kruskall Wallis tests were used.

Sources of variation in logging effects

We pursued several lines of inquiry to explain the observed variability of departure in logged sites. In evaluating this question, we focused our analysis on two response variables, 1) the total density of logged sites and 2) the total density difference between paired logged and unlogged sites, and a number of explanatory variables related to habitat type, local environmental conditions, and the type of logging that occurred. The density difference between logged and unlogged sites was defined as logged site density minus unlogged site density, such that positive values indicated logged sites were denser than unlogged sites and negative values indicated the reverse. For this analysis all logged and unlogged sites from Naficy (Chapter 1) were used for a total of n = 23 pairs. We used the habitat type classifications of Pfister et al (1977) for sites in Montana and that of Steele (1981) for sites in Idaho. Habitat types describe hierarchically nested species association groups ordered by indicator species associated with specific site conditions. Habitat types are useful surrogates of local integrated environmental site conditions, including soil type, moisture availability, elevation and heat exposure. To assess whether either response variable was influenced by habitat type, we conducted an ANOVA analysis with habitat type as a factor and both response variables as dependent variables. Although we sampled sites in ponderosa pine (n=3 pairs), Douglas-fir (n=18 pairs), and grand fir (n=2 pairs) habitat type series, we chose to pool sites from the Douglas-fir and grand fir series because of the similarity of data values between Douglas-fir and grand fir sites and the low number of samples in grand fir series. Therefore, the ANOVA analysis of density variables by habitat type included comparisons between ponderosa pine versus Douglas-fir/grand fir series combined, for each density variable. Assumptions of normality and homogeneity of variance were verified for all variables.

We used the MT-CLIM program (Glassy & Running 1994, Running et al 1987, Hungerford et al 1989) to extrapolate weather and environmental data for each site based on site physiographic data and daily data records from nearby weather stations over the period 1980-2005. Weather station data were obtained online from the National Climatic Data Center (available at http://www.ncdc.noaa.gov/). The MT-CLIM program uses daily records from base weather stations along with site specific physiographic information such as slope, aspect, elevation, latitude and longitude to calculate daily precipitation, vapor pressure deficit (VPD), solar radiation (Srad), maximum daily temperature (Tmax), minimum daily temperature (Tmin), and average daily temperature (Tday). From these outputs, we calculated average daily Srad, VPD, Tmax, Tmin, and Tday across all years for the growing season, defined as April 1st-September 30th. Mean annual precipitation and growing season precipitation were also calculated. Years for which weather records were incomplete were excluded, but these cases were minimal. For each of these variables, annual averages of daily values were then reduced to a single average value for each site over the entire time period 1980-2005.

We employed two metrics, percent logged and logging intensity, to evaluate the effect that differences in logging treatment might have on the variation in logged stand density or in the density differences observed between paired logged and unlogged sites. The percent logged was calculated as a simple proportion of the basal area that was removed by logging to the total historical large tree basal area of the site. We used historical large tree BA rather than total stand BA because only stumps from large and some medium sized trees (range = 30-91 cm DBH, avg. = 55 cm DBH) remained in logged sites. Basal area removed by logging was estimated from residual stumps found within logged sites. The diameter of all stumps was measured at the highest point on the stump where diameter could accurately be measured. The height from the ground to where the diameter measurement was taken was recorded. In order to convert stump diameter measurements to DBH we developed a simple regression model of tree diameter versus tree height. Since all identifiable stumps were ponderosa pine trees, we measured the diameter of 30 ponderosa pine trees throughout our sites at 20 cm intervals from the ground to breast height (137 cm). A regression of diameter versus height of measurement was conducted ($p < 0.05$, $R^2 = 0.025$) to produce a conversion factor that was then used to convert stump diameter at the measured height to DBH. The DBH values for each stump in a site were then converted to basal area measurements ($m^2 ha^{-1}$) and summed, resulting in an estimate of the basal area removed during harvest for each site. Total historical large tree BA in logged sites was calculated as the stump BA added to the BA of all living trees > 40 cm DBH. We then calculated the percent logged for each site by dividing the BA harvested in that site by the total BA of trees > 40 cm DBH in site.

The second metric used to evaluate the effects of logging treatment was logging intensity. Logging intensity was calculated by multiplying the percent logged by the total BA of harvested trees. Logging intensity, therefore, accounts for the percent BA removed from a site as a result of harvest but is weighted by the absolute BA removed from a site. Thus, two sites with equivalent percent logged could have different logging intensities if the total amount of BA removed varied between sites. This was useful in distinguishing between low and high density sites that may have experienced equivalent percent logged. In this case, sparsely treed sites would have a lower logging intensity value than denser sites because the absolute number of trees removed from dense sites was greater.

All environmental outputs from MT CLIM and both logging treatment metrics were then input as explanatory variables in a regression analysis of density parameters. In addition to the total density parameters used in the habitat type analysis we also included 1) density of small trees < 40 cm DBH of all species 2) density of Douglas-fir and all tree species other than ponderosa pine combined. Both of these added parameters were calculated for logged sites and the difference between logged and unlogged sites, resulting in four new added density variables in addition to the two preexisting total density variables. We employed backwards regression techniques utilizing Akaike's Information Criteria to evaluate which environmental and logging variables fit the best model for each of the six density variables. For this analysis all density, environmental and logging treatment variables were tested for normality assumptions and were natural log transformed where necessary.

Results

Stand structure & composition of reference (burned) stands

As described in other studies of reference conditions for forests characterized by frequent disturbance, we found evidence of variable structure and composition of reference, fire-maintained forests. Burned sites were generally dominated by multiple size classes of ponderosa pine mixed with variable numbers of large and medium Douglas-fir trees (Table 5, 2a-b, Fig. 7b-c). Other tree species were absent from most burned sites and observed only in very small numbers when present (Table 5, Fig. 7d). Most often, these other tree species were grand fir. The density of ponderosa pine trees was fairly evenly distributed across size classes, although slightly skewed towards smaller trees (Table 6a, Fig. 7b). Small Douglas-fir and other shade tolerant trees 5-20 cm DBH were present only in low abundance in burned sites (Table 6b, Fig. 7c), supporting the notion that these small trees were excluded by intermittent fire. However, Douglas-fir trees 20-60 cm, especially those between 20-40 cm, were commonly found in moderate numbers in burned sites. Large Douglas-fir (>60 cm) were present in many sites in low numbers (Table 6b, Fig. 7c). Variability of total stand density in burned sites was primarily related to the variation in small ponderosa pine 5-20 cm and Douglas-fir 20-40 cm DBH (Table 6a-b). While some burned sites had high density of small ponderosa pine trees relative to the density of other size classes, other sites were entirely lacking recruitment of any ponderosa pine 5-20 cm DBH (Table 6a). Burned sites had notable variability of large ponderosa pine, with some sites characterized by very low levels of large ponderosa pine trees.

Frequency and degree of departure

Total stand density of many unburned and the majority of logged sites fell outside reference ranges of variability (Fig. 5a. Table 5a). However, unburned sites remained within reference ranges of variability 33-53% (IQR) and 67-83% (range) of the time for Keeling et al (2006) and our sites respectively, whereas logged sites did so only 13% (IQR) and 26% (range) of the time (Fig. 6a). Although most unburned and logged sites showing departures from the reference range of variability were above reference densities (herein termed “positive departures”) a few sites exhibited lower ponderosa pine and

Douglas-fir density than the observed IQR for reference sites (herein termed "negative departures"). However, no negative departures from the full range of reference densities were observed for unburned sites, while 9% of logged sites showed negative departures from the reference range of density (Fig. 5a, Table 5a). Overall, logged sites exhibited notable variation in stand density departures from reference ranges of variation. In general, where negative departures from reference ranges of variation occurred, they were modest compared to positive departures (Fig. 5). On average, positive departures in logged sites were from 2-2.6 times larger than the maximum density observed for the Range and IQR of reference sites, respectively (Fig 2a, Table 5a, b). In contrast, unburned sites which departed from reference ranges of variability showed smaller departures in most cases with an average total stand density departure of 1.5-1.8 and 1.5-1.6 fold for the IQR and Range, of our sites and those of Keeling et al (2006), respectively (Fig 2a, Table 5a, b).

In unburned sites, ponderosa pine density showed occasional departures from the reference IQR and virtually no departures from reference ranges of density, whereas frequent departures were observed in ponderosa pine density of logged sites (Fig. 5b, Table 5a). Where ponderosa pine density in unburned sites did depart from reference ranges of variability, the degree of departure was minimal (Fig. 5b, Table 5b). Logged sites showed similar frequency of negative and positive departures from the IQR, but negative departures were greatly reduced when compared to the full range of reference densities (Table 5a). Positive departures of ponderosa pine density in logged sites were similar to those observed for total density, ranging from 2.2-2.6 fold increases relative to the range and IQR, respectively.

Douglas-fir density was the component of stand structure in unburned sites which exhibited the most frequent departure from reference ranges of variability, departing 67-74% (IQR) and 33-52% (range) of the time for Keeling et al (2006) and our sites, respectively. For logged sites, the frequency of departure in Douglas-fir density was greater than unburned sites, but similar to the departure frequency exhibited for ponderosa pine in logged sites. Whereas ponderosa pine departures in unburned sites were very minimal, departures of Douglas-fir were more substantial, generally upwards of a two fold increase (Fig. 6b). However, the magnitude of departure observed in

Douglas-fir density of logged sites was almost twice that observed in unburned sites, on average, and 4-5.4 times the maximum density observed in reference sites for the range and IQR, respectively (Table 5b). For other tree species, unburned and logged sites showed similar frequency of departure, ranging from 39-50% of sites, independent of which range of variation metric was used (Fig. 5d, Table 5a). In both logged and unburned sites, Douglas-fir and especially other tree species exhibited substantial variation in the magnitude of departure experienced, although due to the low number of other tree species found in reference sites the degree of departure was generally quite large. For both Douglas-fir and other tree species, negative departures were rare (Fig. 5c-d, Table 5a).

Comparison of stand structure & composition between burned, unburned and logged sites

Mean total stand density tended to be higher in unburned stands than burned stands ($p < 0.062$) and was significantly higher in logged stands than in either burned ($p < 0.001$) or unburned ($p < 0.01$) stands (Table 5, Fig. 7a). Compared to burned sites, unburned sites had similar density of large trees > 60 cm DBH of all species (Fig. 7, Table 6). The only component of unburned stands that was significantly different from burned stands was small Douglas-fir trees 5-20 cm DBH ($p < 0.001$) (Fig. 7c, Table 6b). Across all treatments, medium (40-60 cm DBH) and large (> 60 cm DBH) Douglas-fir trees were present in moderate numbers in some sites and absent in others, but these patterns did not vary between treatments (Fig. 7c, Table 6b). No statistically significant differences in ponderosa pine tree density were found between burned sites and unburned or logged sites. Burned and unburned stands had very similar structure of ponderosa pine trees, although unburned sites tended to have somewhat higher density of ponderosa pine 5-20 cm (Fig. 7b, Table 6a).

Relative to burned sites, higher stand density in logged sites was due to small Douglas-fir trees 5-20 cm ($p < 0.001$) and 20-40 cm ($p < 0.01$) DBH (Fig. 7c, Table 6b). Logged sites were dominated by trees < 40 cm DBH, in much higher densities than burned or unburned sites, and had a lower density of large ponderosa pine trees > 60 cm DBH than either burned ($p < 0.094$) and unburned ($p < 0.001$) sites (Fig 1a-d, Table 6a-b). Median density of ponderosa pine trees in logged sites was similar to those of burned and unburned sites (Table 5), but average ponderosa pine density of logged sites was higher than either burned or unburned sites due to the high density of small ponderosa pine in the two smallest size classes found in many sites (Table 6a). This tendency was reflected in high variability of ponderosa pine density for the two smallest size classes in logged sites (Table 6a). Both unburned and logged sites had substantially greater average density of shade tolerant species in the smallest two size classes relative to burned sites (Fig. 7d). However, median density values were much more similar to burned sites (Table 5), reflecting that while some sites experienced dramatic increases of shade tolerant tree density, most departures were minimal. As a result, no statistical differences were found between treatments for any shade tolerant tree species. The variation of density in logged sites was much greater than in burned or unburned sites for most species and size class combinations (Tables 1 & 2).

Sources of variation in response to logging

Total density of logged sites and the density difference between paired logged and unlogged sites were statistically different between sites in the ponderosa pine and Douglas-fir/grand fir habitat types (Fig. 6). Logged sites in the ponderosa pine series had an average density of 50 trees/ha, whereas logged sites in Douglas-fir/grand fir series had 745 trees/ha on average. Furthermore, the average total density difference between paired logged and unlogged sites was -60 trees/ha for sites in the ponderosa pine series and 396 trees/ha for sites in the Douglas-fir/grand fir series, meaning that logged sites in the ponderosa pine series were on average less dense than unlogged sites while the reverse was true for sites in wetter habitat types (Table 6, Fig. 6).

Based on these results and the observations displayed in Fig. 5a-d that stand density of sites in the ponderosa pine series consistently showed either no departures or only mild negative departures from reference ranges of variability, we removed sites in the ponderosa pine series from the regression analysis of environmental and logging variables to explain the variation in density departures of logged sites. Regression models for logged sites were only significant at the $\alpha = 0.05$ level for Douglas-fir and shade tolerant species density and for small tree density (< 40 cm DBH; Table 7a). Only the model for Douglas-fir and other shade tolerant species was significant at the $\alpha = 0.05$

level for the density difference between logged and unlogged sites (Table 7b). However, all models for logged sites and the density difference between logged and unlogged sites were significant at $\alpha = 0.1$ level. Logging intensity was the only explanatory variable which consistently produced the best model fit of all density parameters for logged sites and the density difference between logged and unlogged sites. Along with logging intensity, Tmax was included in the best models for Douglas-fir and other tree species density, although it was only significant ($p < 0.05$) for the density difference between logged and unlogged sites. In all cases logging intensity was positively correlated with density parameters, whereas Tmax was inversely correlated with density parameters. The three significant models (i.e., $p < 0.05$) explained from 19-43% of the variation in the density data (Table 7a, b), suggesting that although logging intensity affects the density of small trees and shade tolerant trees, and thus total stand density, other factors play a significant role in determining the ultimate stand density characteristics of disturbed forests.

Discussion

Increasingly, restoration of altered landscapes has become an important framework for management of federal public lands. With over 10 million hectares of the western U.S. estimated to be in need of restoration (Stephens & Gill 2005), forest restoration treatments are likely to be the singularly most widespread management activity on public lands over the next several decades (Stephens & Moghaddas 2005). On National Forest lands alone, the U.S. Forest Service has set a goal of treating 1.5 million ha/year to reduce fuels and restore historical vegetation composition and structure (GAO 2003). Such a monumental task can only benefit from clear expression of goals and commensurate prioritization of treatments where they are likely to provide the greatest gains toward fulfillment of these goals.

Reference forest conditions in Northern Rockies ponderosa pine forests

The structure and composition of our reference sites is similar to reference conditions reported in the few studies of undisturbed ponderosa pine forests in the Northern Rockies (Arno et al 1995, Arno et al 1997). Based on stand reconstructions of mature unlogged ponderosa pine/Douglas-fir forest, Arno et al (1995) report stand densities of 99-247 (average = 153, median = 127) trees/ha in pre Euro-American settlement (circa 1900) forests of western Montana. This fits within the range of 17-433 trees/ha observed in our burned sites and even more closely matches the IQR, mean and median (Table 5). Likewise, the low proportion of grand fir and moderate representation of medium sized Douglas-fir was similar in both studies. An exception was the density of ponderosa pine estimated by Arno et al (1995) at 46-148 (mean = 72, median = 59) trees/ha, which is substantially lower than the 8-283 (mean = 147, median = 138) trees/ha in our burned sites (Table 5). In both studies by Arno et al, all trees greater than 4.5 feet and 90+ years old (~4 in. dbh) were measured. Trees with germination dates after 1900 A.D. were considered to be a result of fire exclusion and were not included in reference (i.e. 1900 A.D.) conditions. Generally, the density of our reference sites overlaps the broad range of values found in studies of mature or old growth ponderosa pine forests outside the Northern Rockies (Stephens & Fule 2005, Harrod et al 1999, Ehle & Baker 2003, Fule et al 2002, Minnich et al 2000, Stephens & Gill 2005, Fule & Covington 1998), although important differences do exist between regions (Table 8). Regional density variation may be due to differences in climate, microsite conditions, disturbance regimes, or the presence of other conifer or deciduous cohabitants.

An important assumption of our approach in this study is that the sites from Keeling et al (2006), especially the burned sites which we use as our reference sites, are an appropriate comparison for the unlogged and logged sites from Naficy (Chapter 1). Due to the complexity of land management history and the rarity of fire-maintained sites outside wilderness and remote unroaded areas, a three way paired comparison of sites with all three treatments (i.e. burned, unlogged, and logged) was not possible. We therefore depend on the two pronged approach presented here. Such approach has certain limitations, including 1) a limited sample size (n=6) of burned sites to use as our reference baseline and 2) the lack of direct pairing of burned versus logged sites.

Low sample size is a logistical limitation that affects both the statistical power and geographic inference of our reference sites. Statistically, the low sample size likely reduced our power to detect differences in response variables between sites with different land management histories. Likewise, the low sample size may also have affected the accuracy of our measures of variability, as it is unlikely that we captured the full extent of variation that exists in these forest types. We acknowledge this limitation and account for it in part through the use of conservative, non parametric statistical comparisons such as bootstrapping. In Chapter 1, we evaluated assumptions underlying the coupling of our data set with that from Keeling et al (2006) and found no evidence that such comparison was unwarranted due to geographic variation in stand dynamics or response to management history.

We evaluated the validity of geographic inference of sites from Keeling et al (2006) and our study by testing whether statistical differences existed for density parameters between unlogged sites from Keeling et al (2006) and both the full set (n=23) of unlogged sites from Naficy (Chapter 1) and a subset (n=8) of these sites located in the subregion studied by Keeling et al (2006). We found no differences between unlogged sites from each study for either the full analysis or the subset analysis (see Naficy Chapter 1), suggesting that random site effects between sites from the two studies are likely not a significant contributing factor to the observed differences. To further verify that comparisons between sites from both studies are valid, we ran all statistical tests of differences between our logged sites and the pooled unlogged sites from Keeling et al (2006). Results were similar to those found using our unlogged sites alone, suggesting that comparisons between sites from the different studies are valid. Finally, stand density values for all unlogged sites and for burned sites from Keeling et al (2006), closely match values presented by Arno et al (1995) and Arno et al (1997), providing further reassurance that our data sufficiently represent stand conditions associated with particular land use histories.

<u>Assessing the need for restoration of unburned, unlogged and unburned, logged forests</u>

Although our results corroborate the general finding that lack of frequent fire and logging have both contributed to departures from reference ranges of variability, we document important differences in the patterns and magnitude of departures experienced by logged, unburned and unlogged, unburned forests. Our data provide strong evidence that distinct restoration approaches are warranted for sites with these different land management histories. Contrary to popular assertion that fire exclusion has caused the most ubiquitous and dramatic changes in forest structure (Agee & Skinner 2005, Covington 2000), the analyses presented here and in Keeling et al (2006) suggest that fire exclusion effects in ponderosa pine forests of the Northern Rockies, while significant, are significantly less than fire exclusion effects in combination with past logging. While it is clear that fire exclusion has the general effect of moving ponderosa pine forests of the Northern Rockies towards late successional conditions (Keeling et al 2006, Arno et al 1995, Arno et al 1997), many of these forests are not necessarily outside the range of conditions found in more frequently burned reference sites (Fig. 5). In fact, we found that despite more than 65 years since the last fire occurred in our unburned sites many of them (up to 83%) are still within the observed ranges of variability of burned sites (Table 5a) and even those which have experienced departures have generally done so only to a mild degree (Table 5b).

Our data clearly demonstrate that the combined effects of logging and lack of fire have produced the most consistent and dramatic departures from reference conditions. Using the IQR as a conservative estimate of the reference range of variability, only three logged sites did not depart from the reference ranges of variability at all and, surprisingly, two logged sites showed significant negative departures from even the minimum stand density observed in reference sites. The latter occurred in logged sites that had minimal to virtually no successful recruitment following harvest. These sites consisted of open, grassy patches with many visible stumps and an occasional small pine or Douglas-fir tree. These exceptions aside, logged sites were consistently dominated by a homogeneous profusion of small Douglas-fir, ponderosa pine and other shade tolerant trees < 40 cm DBH.

Consistent with other studies of fire-maintained forests in this region (Arno et al 1995, Arno et al 1997), our data show the upper limit of total stand densities observed in burned stands to be 308 trees/ha based on the IQR or 433 trees/ha for the maximum observed density. In the absence of additional data, these values may serve as useful reference values for ponderosa pine/Douglas-fir forests of the Northern Rockies from which to estimate departures of stand density as we did here. However, it should be noted that even within the Northern Rockies bioregion, our reference sites may not accurately represent reference ranges of variability for the driest ponderosa pine forests, such as those found in the ponderosa pine habitat type series, or mesic ponderosa pine forests where species such as western larch (*Larix occidentalis*) and lodgepole pine (*Pinus contorta*) are intermingled in substantial abundance. Where stands are deemed to be outside the range of variability, we outline below some prescriptive guidelines for restoration of unlogged, unburned and logged, unburned stands. Our recommendations are based in the perspective that the most parsimonious vegetative prescriptions which restore forest structure and composition to reference ranges of variability is the most favorable. This suggestion is made in light of three recognitions: 1) our understanding of historical conditions is often incomplete, especially with regards to the small tree component which can vary greatly and can contribute greatly to total stand density 2) the restoration of biological processes is paramount to successful restoration and restoration of structural and compositional components of an ecosystem are necessary primarily to allow these processes to regain functionality and 3) vegetative restoration treatments themselves have negative impacts, and unknown risks. Negative impacts of tree cutting for restoration include: potential long-term density feedbacks associated with logging disturbance, increased weed spread, compaction of soils, wildlife disturbance, the need to build new roads or reconstruct old revegetated roads, and slash creation and increased wildfire hazard (Rhodes & Baker 2008, Naficy Chapter 1, Nelson et al 2008). Many of these are a direct result of the heavily industrial nature of modern silviculture, and could be avoided if alternative tree cutting methods or stringent restrictions are placed on when, where and how implementation occurs.

Restoration prescriptions for unburned, unlogged sites

There is evidence that despite structural changes associated with a lack of fire, many unlogged, unburned forests are not in need of mechanical treatment before the reintroduction of natural wildfires (Collins & Stephens 2007, Fule & Laughlin 2007, Odion & Hanson 2006, Holden et al 2007). While this may not universally be the case (Goforth & Minnich 2008), it appears that in many cases the proactive allowance of natural wildfires may be the most effective and parsimonious form of ecological restoration of semi-arid western U.S. forests (Collins & Stephens 2007, Fule & Laughlin 2007, Odion & Hanson 2006, Holden et al 2007). The aim of the following prescriptive recommendations for mechanical manipulation of stand characteristics is not to guide restoration towards static vegetative reference targets, but instead aims to restore adequate semblance of the reference structure and composition so that fire may function relatively unimpeded by past human influence. For these reasons, in combination with the known and unknown risks associated with mechanical treatments, we preface our recommendations for mechanical stand manipulations of unlogged, unburned ponderosa pine forests of the Northern Rockies with these caveats and frame them as upper possible limits rather than necessary minimum thresholds. The point cannot be overemphasized, however, that without the reinstatement of natural wildfires, or application of prescribed fires of similar intensity and frequency, the value of these prescriptions as a form of ecological restoration is highly questionable.

Unburned sites generally retained all of the structural features of burned sites, and had departed from reference ranges of variability only in the extent to which small Douglas-fir trees 5-20 cm DBH and, in some sites, shade tolerant trees 5-40 cm DBH (Fig. 7c-d, Table 6) were present. Restoration treatments of these sites should emphasize the removal of these small, shade tolerant trees. All ponderosa pine trees including small recruiting trees 5-20 cm should be preserved. Contrary to findings in southwestern ponderosa pine forests, where fire exclusion has led to a profusion of ponderosa pine recruitment, our results suggest that fire exclusion diminishes successful establishment of young ponderosa pine trees (Fig. 7b, Table 6a). These data are consistent with predictions made by Baker et al (2007) for ponderosa pine forests of the Rocky Mountains characterized by variable severity fire regimes. Our data suggest that all large and medium sized Douglas-fir should also be preserved. In fact, our data show little evidence that any Douglas-fir > 20 cm DBH should be harvested to restore reference stand conditions in previously unlogged, unburned sites (Table 6b).

Prescriptive vegetation restoration recommendations made in other studies of dry coniferous forests vary in the specific tree harvest prescriptions made, but they show general parallels with our recommendations in emphasizing the removal of small trees, especially of shade tolerant species when present, while retaining large trees. Differences in the threshold tree sizes recommended for treatment may vary due to regional differences in site productivity and resultant average tree size and density, regional variation in commencement of effective fire exclusion, or differences in the approach to or goals of restoration treatments. Covington et al (1997) recommended retention of all presettlement trees and trees > 16 inches (40 cm) DBH, with retention of 3 postsettlement trees per presettlement tree, preferably arranged in clumps, to serve as a recruitment cohort. Thinning treatments in their "full restoration" treatment were then followed by prescribed burning in the late fall, with subsequent repeated burns at historical fire rotation intervals. Moore et al (1999) describe the approach developed for southwestern ponderosa pine forests in greater detail. North et al (2007) studied mixed conifer forest in the Teakettle Experimental forest of the central Sierra Nevada range in California. They state that "effective treatments should drastically reduce densities of small trees (<50 cm DBH), retain some intermediate-sized and all large trees, significantly decrease the percentage of white fir, and reduce stem clustering." In mixed conifer forests of southern California, Goforth & Minnich (2008) found that fire exclusion had allowed conifer density to increase from 271±82 to 716±79 trees ha^{-1}, primarily due to saplings and pole-sized (10-29.9 cm DBH) stems of shade tolerant *Calocedrus decurrens*. They imply that it was these *C. decurrens* stems 10-29.9 cm DBH that enabled an aberrant stand replacement fire of almost the entire 4,000 ha mixed conifer forest on Cuyamaca Mountain in southern California. In mixed ponderosa pine/Douglas-fir stands of western MT near many of our sites, Arno et al (1995) and Arno et al (1997) recommend that understory trees be heavily thinned and that canopy openings be created to stimulate regeneration of ponderosa pine.

Another common modern silvicultural recommendation that is widely discussed as a useful restoration tool is the harvest of some medium and large trees to reduce canopy continuity and create small canopy openings. Such treatments are intended to make forests more resistant to crown fire (Stephens & Moghaddas 2005, North et al 2007, Agee & Skinner 2005, Graham et al 1999) and to augment regeneration of shade intolerant tree species such as ponderosa pine (North et al 2007, Zald et al 2008, Arno et al 1995). Our data do not support this practice as a valid form of ecological restoration for previously unlogged ponderosa pine forests of the Northern Rockies. Removal of trees > 40 cm DBH would not be consistent with the types of departures experienced by unlogged, fire excluded forests in our region (Fig. 7, Table 6) and would reduce the density of ecologically valuable and rare large trees. Notwithstanding the exclusion of canopy removal treatments, it is important to recognize that our recommendations will confer many of the benefits of reduced crown fire risk in frequent fire forests, consistent with general principles of vegetation treatment effects on fire behavior (Agee & Skinner 2005, Graham et al 2004). Primarily, this can be accomplished by reducing surface fuels through prescribed burning, increasing height to live crown by thinning small trees < 20-40 cm DBH, depending on species, and by maintaining large, fire-resistant structures.

<u>Restoration prescriptions for logged sites</u>

Logged, fire excluded sites lack much of their historical stand structure and exhibit great variability, which complicates the ability to provide simple restoration prescriptions. Restoration treatments should reduce the density of small trees < 20 cm DBH of all species, although an adequate number of small ponderosa pine commensurate with the range observed in burned reference sites should be retained (Table 6a). Douglas-fir 5-20 cm DBH and other shade tolerant species < 40 cm should be targeted by thinning prescriptions to return them to near reference levels (Table 6b). To restore some portion of the large tree component that characterizes mature sites, treatments should preserve all ponderosa pine and Douglas-fir > 40 cm DBH.

Although it has been cautioned that leaving overstory shade tolerant trees can contribute to local inertia against the reestablishment of a dominant overstory of shade intolerant trees (Zald et al 2008), we found that Douglas-fir > 40 cm DBH in logged sites were well within, or slightly below, the range of observed densities in burned reference sites and therefore should not be harvested (Table 6b). Only two logged sites had any trees other than ponderosa pine or Douglas-fir > 40 cm DBH (data not shown), and these were found in low numbers (10-20 trees/ha). Whether these trees should be retained to augment the depauperate large tree component of logged sites or removed to limit seed source of shade tolerant trees is unclear. However, the issue of seed rain effects by shade tolerant tree species may be a mute point if wildfire is allowed to return, or prescribed fire is applied periodically. In this case, it is likely that many shade tolerant trees, including medium and larger diameter trees, would be killed (Hood et al 2007), thus limiting their contribution to seed rain and providing a valuable habitat to snag-dependent species in stands that are otherwise severely lacking in large snags.

Difficulty in defining restoration targets for logged forests

Depending on the degree of residual structure remaining in logged sites, using unlogged mature or old growth forests such as we did with our reference stands may be inappropriate (Baker et al 2007). Where significant residual structure remains following harvest, restoration treatments that move stand conditions toward the range of variability of mature or old growth conditions may be feasible. In contrast, where a stand has been largely reinitiated by past silvicultural treatments and little residual structure exists, a more appropriate restoration target might be younger, unaltered stands of similar age. Many forest ecology studies examine stand structure and fire history of old growth or mature stands and emphasize the importance of fire in maintaining forests dominated by large, widely spaced old trees (Arno et al 1995, Stephens & Gill 2005, Youngblood et al 2004, Fule et al 2002). However, few studies of stand structure, composition, fire history, fire effects and fire behavior in younger (40-150 years old) forests have been published, making their use as references a difficult task (but see Ehle & Baker 2003, Sherriff & Veblen 2006).

Whereas mature, fire excluded stands were previously shaped by the recurrent presence of fire at natural historical intervals for 100 to several hundred years, young stands that have been greatly altered by past logging have developed during the fire exclusion era and may have never experienced fire before. Fire exclusion in young stands may therefore have produced much more severe effects than it does in mature or old growth stands (Holden et al 2007). This may be part of the explanation for the much denser conditions often found in logged and fire excluded stands versus unlogged, fire excluded stands (Naficy Chapter 1). While unlogged, fire excluded stands maintain all, or a substantial portion of their original, fire-influenced structure, logged forests have largely lost this past influence of fire on stand structure and have developed their structural attributes in the continual absence of fire. In conjunction with the research presented here, data that address the above uncertainties would provide a critical basis for restoration treatments of ponderosa pine forests of the Northern Rockies.

Variation in logging effect

Few studies have investigated the causes of observed variation associated with managed landscapes. However, Minnich et al (1995) did find a greater positive density response to fire exclusion in wetter sites than drier sites, a likely result of greater recruitment success in wetter sites (Zald et al 2008). In parallel with this finding, we hypothesized that departures in stand density of logged sites from reference ranges of variability would be greater in more mesic sites than drier sites. We found partial support for this hypothesis, as evidenced by the clear divergence of logging effects on density in drier ponderosa pine versus wetter Douglas-fir/grand fir habitat types. Not only did logged sites in the drier ponderosa pine series have much lower density than those in Douglas-fir sites, but the density difference between adjacent logged and unlogged sites was consistently negative for sites in the ponderosa pine series (Fig. 6, Table 6), resulting in all three of these sites showing clear negative departures from reference ranges of variability (Fig. 5). Our study design did not include a large enough sample size of sites in ponderosa pine habitat types to understand the causal mechanisms behind this phenomenon. However, we postulate that despite the passage of 50-100 years since logging occurred in these sites, consistently harsh site conditions have limited successful tree recruitment. Sites in the ponderosa pine series were generally lower elevation sites (elevation range = 976-1,086 m), sometimes occurring at ecotones with grasslands, but generally consisting of open, sparsely treed stands (Fig. 7a). In these sites, it appears that logging has long-term depressive effects on stand density.

In stark contrast, logging had the opposite effect in Douglas-fir/grand fir habitat types (Fig. 7b), resulting in positive density responses that increased in parallel with logging intensity. Logging intensity predicted the density of Douglas-fir and other tree species better than either total stand density or the density of small trees < 40 cm dbh (Table 7), with stronger density feedback effects in cooler sites (i.e. sites with lower Tmax) where Douglas-fir and other shade tolerant species are more likely to occur. Neither precipitation nor vapor pressure deficit improved any of the final density models, implying that even within the Douglas-fir habitat type, which includes a very broad range of site conditions, moisture availability is less important than the intensity of logging disturbance in determining the stand density response to disturbance. While broad differences in site conditions may alter the density response to logging, this relationship is non-linear and appears to be less significant, for all but the driest sites, in driving post-harvest density responses than is the intensity of the disturbance itself.

These results parallel those of Keeling et al (2006) who found that variability of current stand structure in unlogged, unburned ponderosa pine/Douglas-fir stands is not strongly related to environmental or other physiographic site variables. Instead, they suggest that differences in forest structure and composition are most likely related to differences in burn severity of past fires or stochastic differences in initial stand characteristics. All of the sites studied in Keeling et al (2006) were within the Douglas-fir and grand fir habitat type series. Taken together, our findings suggest that except for ponderosa pine forests in the most arid sites (i.e. those in ponderosa pine habitat types), disturbance history, and specifically the severity of past disturbances, is more important in shaping stand structure and composition than are local environmental or physiographic factors. While environmental and physiographic factors may interact with individual disturbance events and influence the nature of broader disturbance regimes for an area, it is primarily in this indirect sense that environmental and physiographic factors shape forest structure and composition in disturbance-mediated systems, rather than influencing the rate or nature of post-disturbance recovery.

In addition to the intensity of a disturbance, the time since disturbance is also known to be an important factor in determining ecosystem response. While we accounted for logging intensity in our analysis, we could not control for time since disturbance (we did not have detailed fire history data for our sites and few detailed records of logging treatments exist). Severe stump decay and heartrot prevented the effective use of dendrochronological techniques for crossdating cores or sections from stumps to determine date of harvest. From what general information we have of harvest dates for each site, it does not appear that time since logging disturbance is a significant factor in determining modern stand density, but we cannot ascertain this, statistically, with the data we collected. It is possible therefore, that part of the logging effect we report is due to stand age effects resulting from the initial intensity of logging and the time since the logging event occurred. Stand data from unlogged, post-fire stands of similar age to our logged sites would be a valuable complement to our data set.

Management implications of controls on variability

The finding that logging intensity is positively correlated with subsequent stand density in unburned sites brings to the forefront the question of what the long-term consequences will be as a result of modern silvicultural prescriptions designed for fuel reduction, restoration, or commercial timber production. If modern silvicultural treatments mimic historical logging practices, and especially if current fire suppression efforts are not widely scaled back, then they may result in similar counterproductive trends in stand density over the course of several decades. While there is a natural tendency to differentiate between the effects of historical and modern logging practices, this distinction is not necessarily warranted and should not be assumed without further study. This point is especially important, given the current lack of long-term monitoring data of fuel reduction and restoration treatments (Brown et al 2004, Pollet & Omi 2002) and the concurrent federal and state emphasis on these treatments as embodied in policy direction such as the Healthy Forests Restoration Act (HFRA 2003), the National Fire Plan (USDA-USDI 2000), and the 10-year comprehensive strategy (WGA 2001).

While there may be significant differences between historical and contemporary silvicultural treatments, it is often the case that modern fuel reduction and restoration projects still depend on harvesting medium and some large trees (Zald et al 2008, Stephens & Moghaddas 2005), both as a method of attaining structural and compositional targets of more "fire resilient" forests and in order to generate revenue to cover project expenses. Detailed historical records describing the silvicultural prescriptions used in our logged sites are unavailable, constraining our ability to deduce the specific logging treatments to information derived from residual stumps. While differing silvicultural prescriptions were likely used in our logged sites, they all involved some degree of overstory tree removal (% basal area of trees > 40 cm DBH harvested, avg. = 71%, median = 71%, range = 19-100%). Such harvests are certainly employed in some widely used modern silvicultural practices. For instance, medium and large trees are harvested in fuels treatments designed to reduce fire spread through the forest canopy, as these treatments expressly depend on removal of overstory trees to attain specific canopy spacing or bulk density targets (Graham et al 2004, Agee & Skinner 2005, Stephens & Moghaddas 2005). Indeed, since our field studies were conducted in the summer of 2005, 56% of our unlogged sites (13 out of 23) have been included in vegetation management projects where some proportion of the trees > 40 cm DBH will be harvested.

While our data do not identify the specific mechanism by which logging causes positive density feedbacks, they do suggest that minimizing logging intensity of medium and large trees in mesic habitat types may help mitigate them, at least in part, and reduce the degree of departure experienced in these sites as a result of treatments. This should be useful in guiding current silvicultural practices which depend on tree harvests to reduce stand density for either fuel reduction or restoration purposes. It is important to note, though, that many logged sites with low logging intensity values still exhibited greater positive density departures than many unlogged, fire excluded sites. This indicates that although logging effects on density may be minimized to some extent, it is likely not possible to avoid significant, long-term positive departures that result from harvest of medium and large trees in the absence of fire by controlling logging intensity alone. However, our regression analysis uncovered a potentially interesting distinction between logging intensity and the percent basal area logged that may offer some insight, albeit limited, into the mechanism associated with positive density responses following logging. That the percent basal area logged was a consistently poor predictor of stand density parameters, while logging intensity was generally a useful predictor indicates that the absolute basal area removed is as, or possibly more, important than the proportion of total stand basal area harvested. This would seem to suggest that the stand density response to logging is related more to ground disturbance than canopy opening. If this should prove to be true as a general rule, it is possible that even the parsimonious recommendations for vegetation treatment we have proposed here could result in positive density feedbacks.

Repeated use of natural or prescribed fire in conjunction with logging treatments may be the most effective method for minimizing positive density feedbacks associated with mechanical vegetation treatments. However, even this proposal is speculative at this point, as recent studies of short-term regeneration following thinning and burning treatments suggest complex, unexpected responses by different species (Zald et al 2008) and there is little long-term monitoring data of the effectiveness of such treatments in predictably structuring patterns of regeneration. Ultimately, more research on the mid- and long-term effects of harvesting understory trees and on the effectiveness of prescribed fire in countering density feedbacks would complement this work greatly.

Conclusions

Stephenson (1999) highlights an interesting and still ongoing divide within the ranks of forest ecologists between "structural restorationists" who assert that mechanical treatment of forest structure and composition is necessary before natural processes, especially wildfire, can be reintroduced without ill effect and "process restorationists" who see mechanical vegetation treatments as potentially harmful and generally unnecessary prior to the reintroduction of natural processes. Our findings neither wholly support nor reject either approach, but instead suggest that the approach is best defined by the specific case at hand and the land management history which has been most influential in shaping it. The minimal departures experienced by most of our unlogged, unburned sites, and the growing evidence that many contemporary wildfires in unlogged, unburned forests do not cause aberrant proportion of high severity fire (Collins & Stephens 2007, Fule & Laughlin 2007, Odion & Hanson 2006, Holden et al 2007) are convincing evidence that a process restorationist approach may be useful for many such forests. In contrast, landscapes characterized by more severe departures resulting from fire exclusion alone or in combination with past logging may benefit from a structural restorationist approach which targets departed vegetation components as we have outlined here, followed by reinstatement of wildfire. This distinction should be welcome news to those interested in forest restoration, as it should help narrow the otherwise Herculean task of treating 10 million hectares of forestland by identifying priority areas for thinning treatments and proposing alternative methods to the costly, controversial, risky, and slow process of mechanical thinning in areas that are in less need of them.

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Figures & Tables

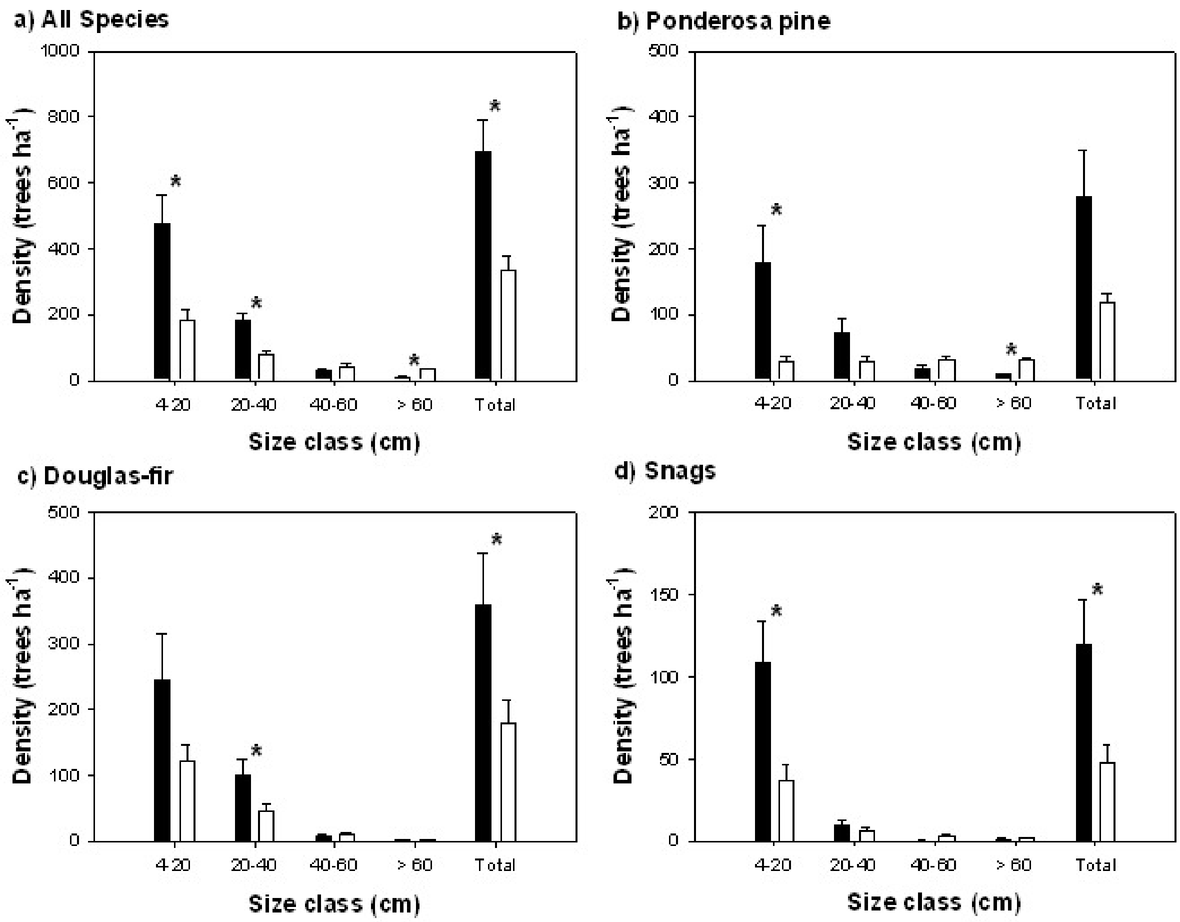

Figure 1. Filled bars represent logged sites and open bars are unlogged sites. Asterisks indicate statistically significant differences between logged and unlogged stands within a size (diameter) class. Bonferroni corrections were used to establish a significance threshold of $p \leq 0.01$ for all comparisons between logged and unlogged stands. Error bars represent ± 1 SE. Size class distribution of density for a) all species, b) ponderosa pine, c) Douglas-fir, and d) snags

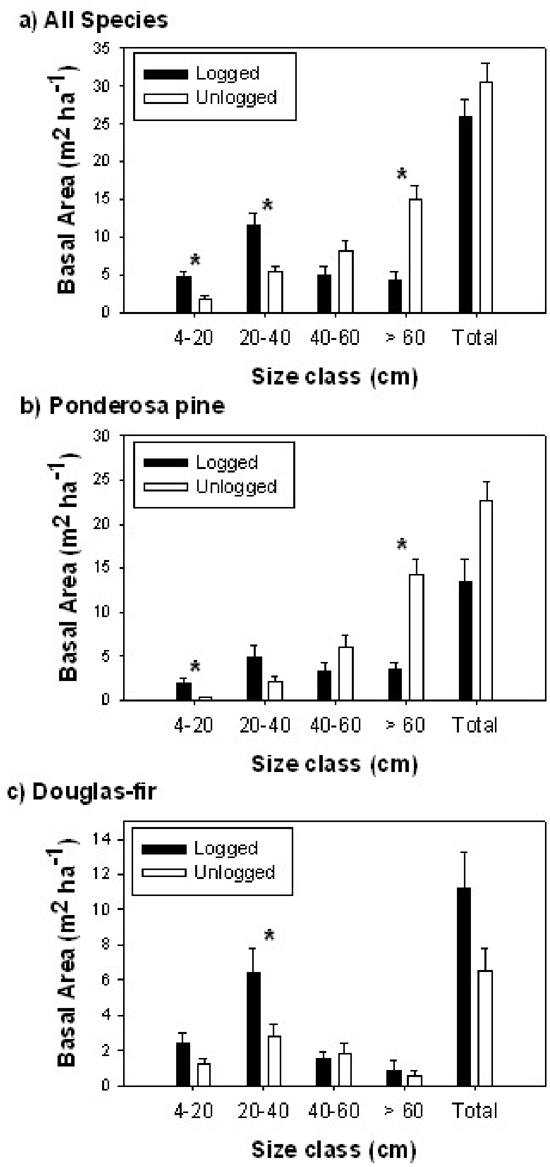

Figure 2. Size (diameter) class distribution of basal area for a) all species, b) ponderosa pine, and c) Douglas-fir. Filled bars represent logged sites and open bars are unlogged sites. Asterisks indicate statistically significant differences between logged and unlogged stands within a diameter class. Bonferroni corrections were used to establish a significance threshold of $p \leq 0.01$ for all comparisons between logged and unlogged stands. Error bars represent ± 1 SE.

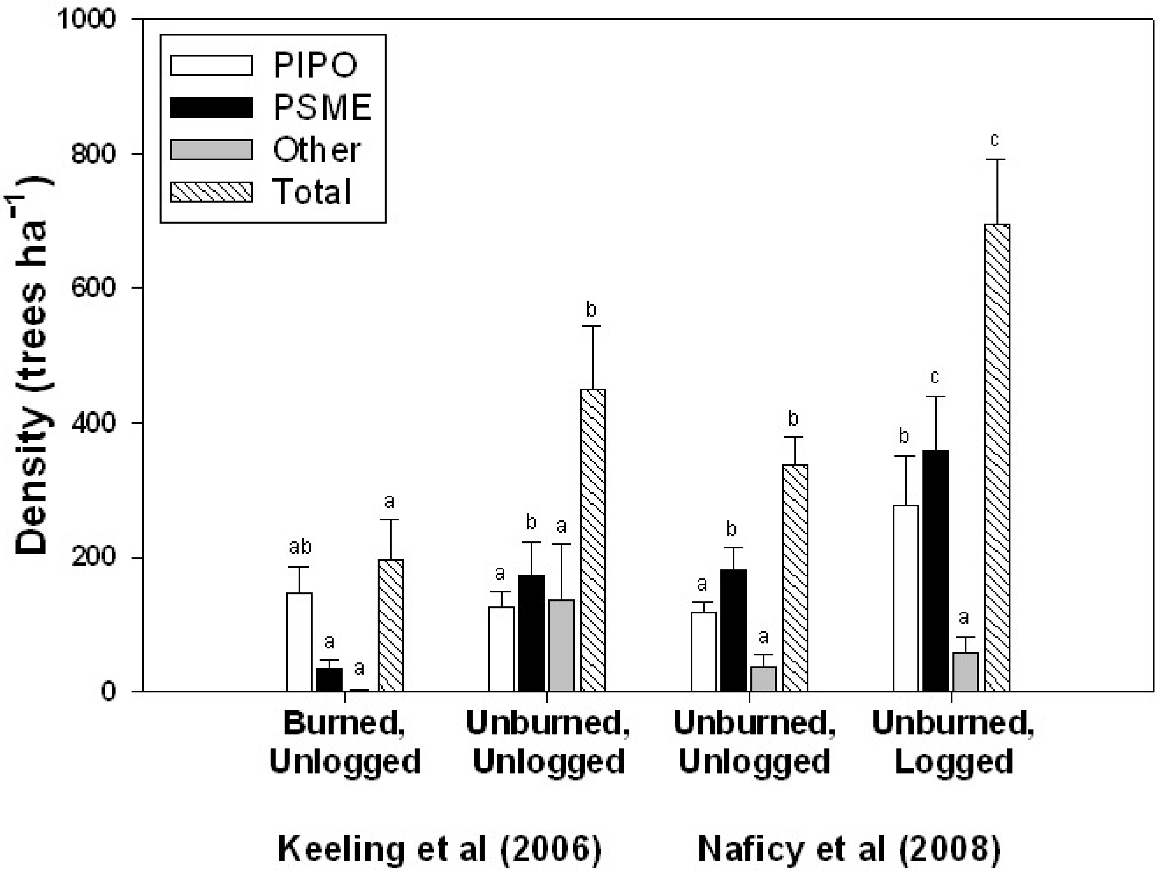

Figure 3. Combined data from Keeling et al (2006) and our study showing tree density for ponderosa pine (PIPO), Douglas-fir (PSME), other species, and all species pooled for unlogged, fire-maintained stands, unlogged fire excluded stands and logged, fire-excluded stands. Within a species, bars with different letters across treatments are statistically significant ($p < 0.05$). Values for all variables in unlogged, fire excluded stands from Keeling et al (2006) and Naficy (Chapter 1) were not statistically different and were pooled together for statistical calculations, although data are shown here separately. Error bars represent ± 1 SE.

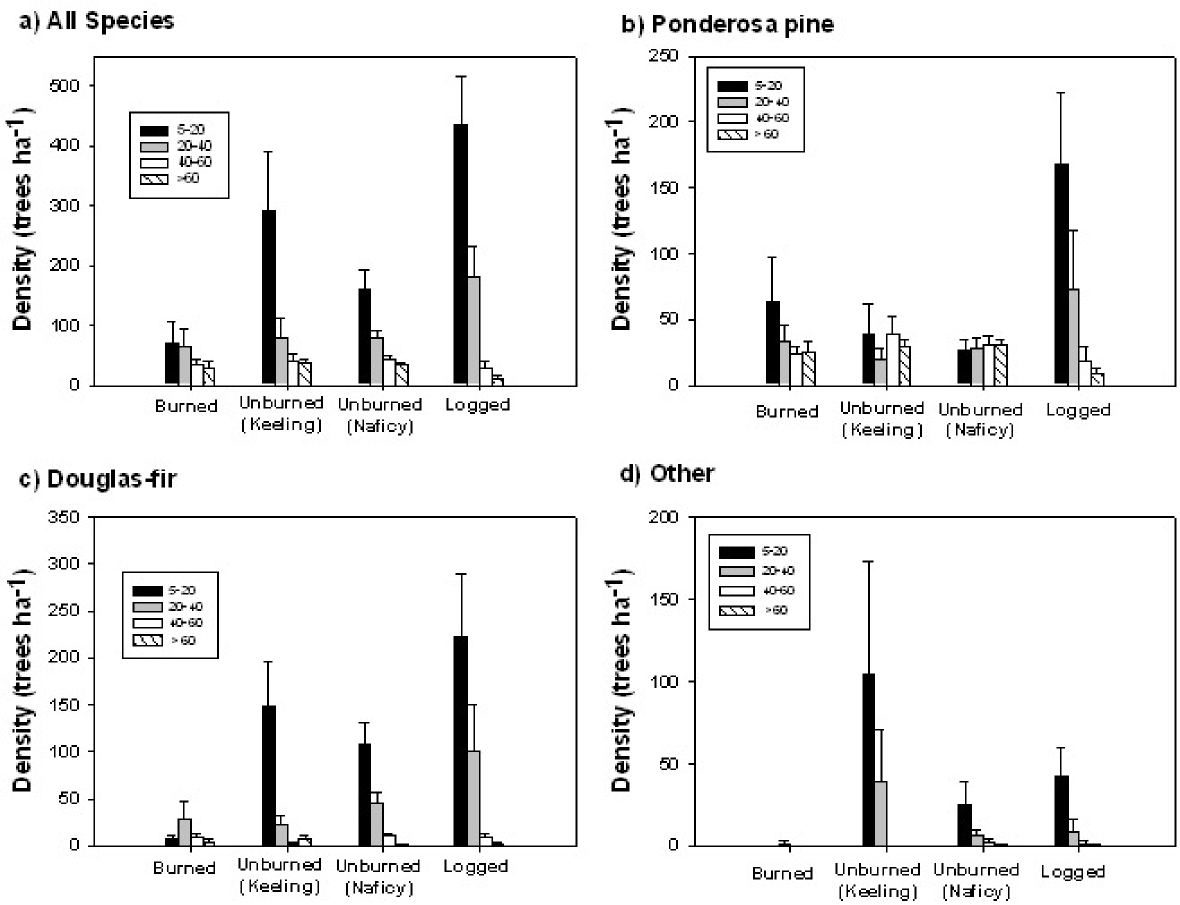

Figure 4. Diameter class distribution for a) all species b) ponderosa pine c) Douglas-fir d) other species for fire-maintained, unlogged and logged stands. Error bars represent ± 1 SE.

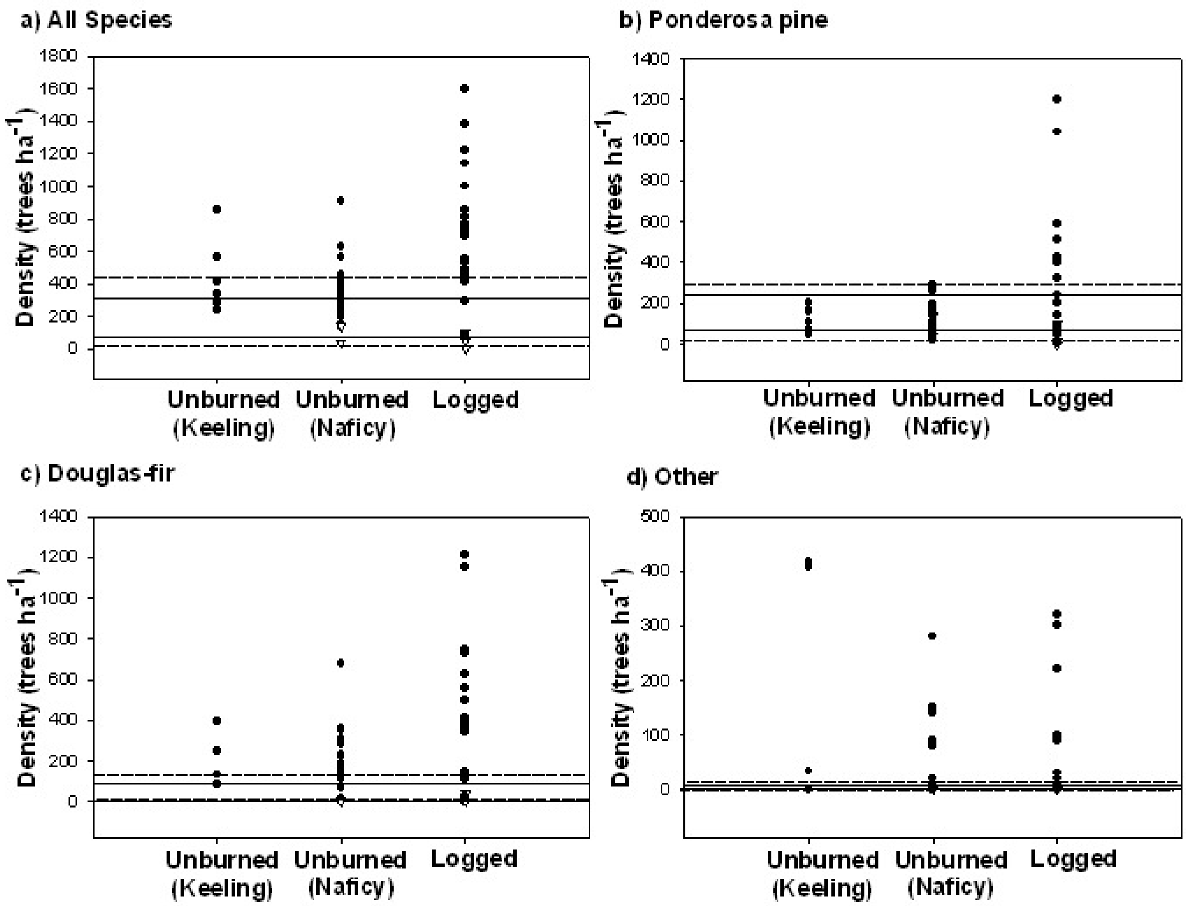

Figure 5. Total density of unburned and logged sites relative to the observed range of variation in burned stands for a) all species b) ponderosa pine c) Douglas-fir d) other species. Solid lines represent the upper (75th percentile) and lower (25th) percentile bounds of the observed interquartile range (IQR) for burned sites. Dashed lines represent the upper (maximum) and lower (minimum) bounds of the full range of values observed for burned sites. Filled circles represent sites within the PSME habitat type series while open triangles represent sites in the PIPO habitat type series.

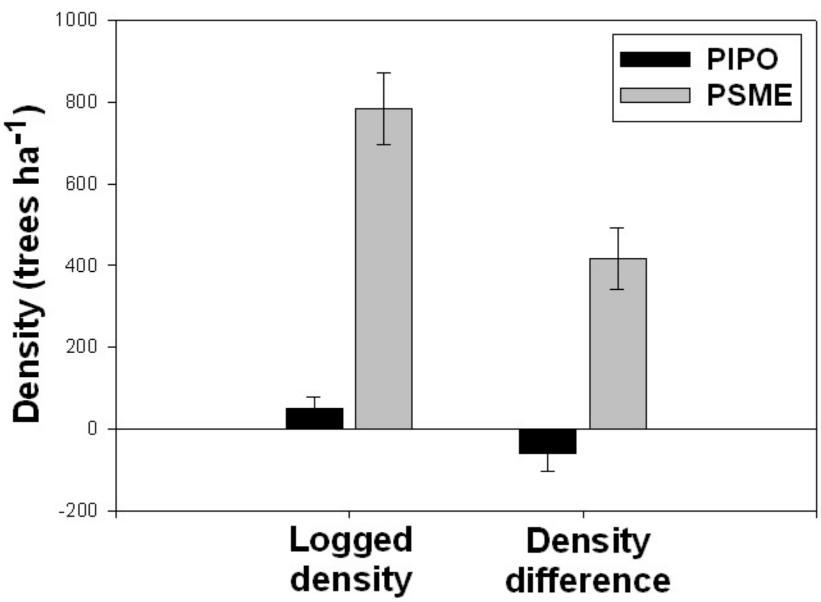

Figure 6. Density of logged sites and the density difference between logged and unlogged sites for ponderosa pine and Douglas-fir/grand fir habitat types. Density difference is calculated as logged – unlogged density. Bars represent ± 1 standard error.

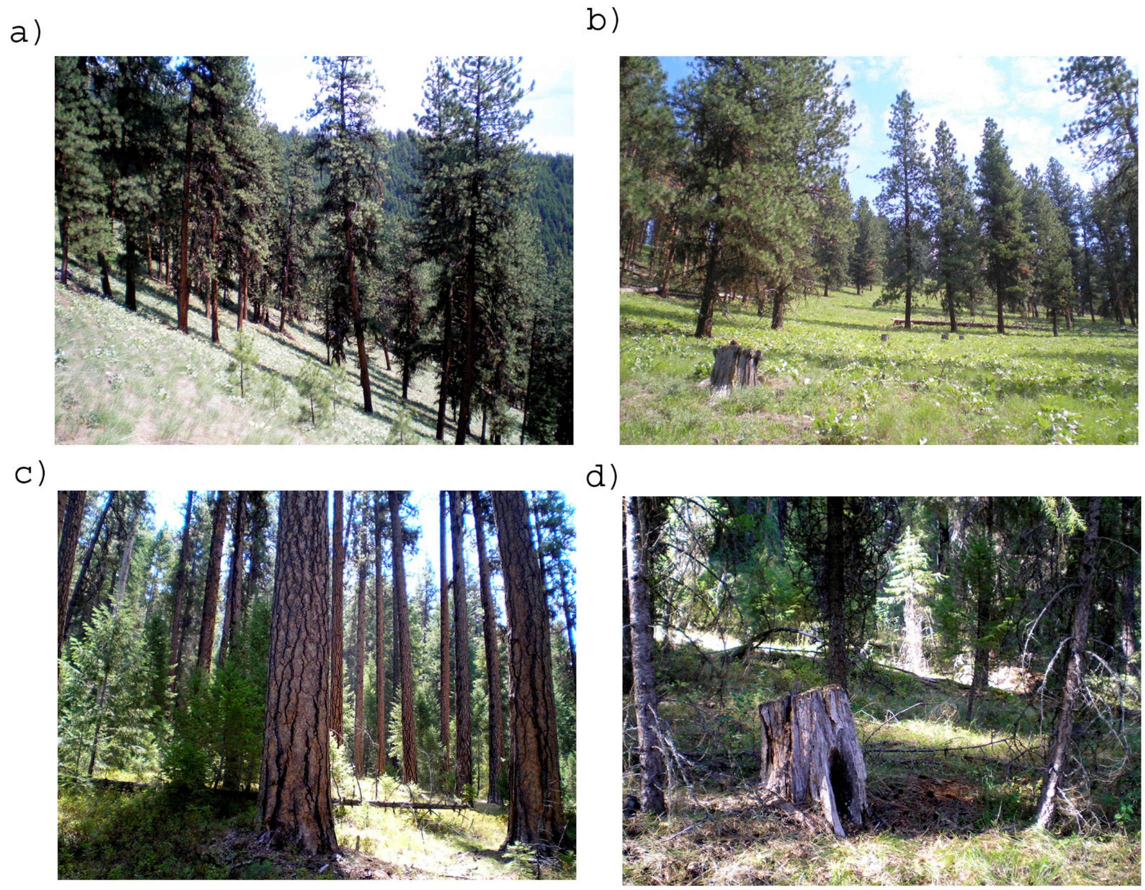

Figure 7. Examples of unlogged, fire excluded (a, c) and logged, fire excluded (b, d) sites in ponderosa pine (a, b) and Douglas-fir (c, d) habitat types. Stumps are visible in both logged sites, but the difference in density response between the two habitat types is apparent. Less dramatic, but also of note is the difference in understory density of unlogged, fire excluded sites.

Table 1. Mean values (standard error) and range for density and basal area of ponderosa pine, Douglas-fir, all species pooled, and snags by size (diameter) class. The percent of total stems < 40 cm DBH was also calculated for each row.

	4-20 (cm)		20-40 (cm)		40-60 (cm)		>60 (cm)		Total		% of Stems < 40 cm	
Density (trees/ha)	Logged	Unlogged	Logged	Unlogged	Logged	Unlogged	Logged	Unlogged	Logged	Unlogged	Logged	Unlogged
Ponderosa pine	178.7* (56.8) 0-1070	28.7 (7.7) 0-130	72.6 (20.9) 0-350	28.3 (7.6) 0-160	18.7 (5.1) 0-90	30.4 (6.5) 0-130	8.3* (2.1) 0-40	30.9 (4.0) 0-70	278.3 (70.0) 0-1280	118.3 (14.2) 20-300	90	48
Douglas-fir	245.7 (68.7) 0-1230	122.2 (24.2) 0-430	101.3* (22.5) 0-300	45.7 (11.5) 0-230	8.7 (2.2) 0-40	10.4 (3.0) 0-30	1.7† (1.0) 0-20	1.7 (0.8) 0-10	357.4* (80.5) 0-1280	180.0 (34.0) 0-690	97	93
All Species	473.9* (86.2) 0-1380	179.6 (33.9) 0-590	182.2* (23.3) 0-350	80.4 (11.6) 10-240	28.3 (5.6) 0-110	43.0 (6.5) 0-130	10.4*‡ (2.4) 0-40	33.5 (4.0) 0-70	694.8* (94.1) 0-1700	336.5 (41.4) 40-920	94	77
Snags	108.3* (25.6) 0-450	36.5 (9.9) 0-190	10 (3.3) 0-60	6.5 (1.7) 0-30	0.4† (0.4) 0-10	3.5 (1.0) 0-10	1.3† (0.7) 0-10	1.7 (0.8) 0-10	120.0* (26.5) 0-470	48.3 (10.7) 0-200	99	89
Basal Area (m^2/ha)												
Ponderosa pine	1.97* (0.62) 0-10.99	0.30 (0.09) 0-1.54	4.77 (1.40) 0-20.41	2.17 (0.56) 0-11.42	3.36 (0.96) 0-18.35	6.00 (1.31) 0-27.33	3.43* (0.91) 0-14.71	14.24 (1.83) 0-28.14	13.52 (2.44) 0-42.42	22.70 (2.12) 7.92-57.43	50	11
Douglas-fir	2.37 (0.64) 0-11.57	1.26 (0.29) 0-5.09	6.38* (1.38) 0-19.59	2.82 (0.69) 0-14.32	1.58 (0.37) 0-5.88	1.85 (0.58) 0-7.81	0.86† (0.60) 0-13.33	0.59 (0.28) 0-4.42	11.19 (2.12) 0-28.53	6.52 (1.25) 0-23.73	78	63
All Species	4.72* (0.82) 0-13.67	1.81 (0.36) 0-6.84	11.69* (1.50) 0-23.32	5.37 (0.69) .32-14.73	5.05 (1.02) 0-21.77	8.28 (1.34) 0-27.33	4.41* (1.10) 0-16.76	15.13 (1.84) 0-28.34	25.87 (2.49) 0-53.88	30.59 (2.54) 7.92-59.8	63	23
Snags	0.63* (0.15) 0-2.32	0.26 (0.08) 0-1.41	0.53 (0.16) 0-2.62	0.39 (0.11) 0-1.65	0.09† (0.09) 0-2.01	0.07 (0.22) 0-2.79	0.65† (0.41) 0-8.59	0.78 (0.02) 0-7.01	1.90 (0.47) 0-8.61	2.14 (0.42) 0-7.32	61	30

* in Logged column indicates significant differences between logged and unlogged stands within a size class. Bonferroni corrections (0.05/5) were used for each row to establish a significance threshold of $p = 0.01$.

† indicates non parametric Wilcoxon signed rank test used

‡ indicates ln transformation

Table 2. P-values for paired t-test comparisons of logged and unlogged stands for ponderosa pine, Douglas-fir, all species combined, and snags by size (diameter) class and total.

Density (trees/ha)	4-20 (cm)	20-40 (cm)	40-60 (cm)	>60 (cm)	Total
Ponderosa Pine	**0.009**	0.027	0.205	**0.000**	0.025
Douglas-fir	0.040	**0.009**	0.657	1.00†	**0.010**
All Species	**0.000**	**0.000**	0.129	**0.001‡**	**0.000**
Snags	**0.002**	0.336	0.020†	0.705†	**0.003**
Basal Area (m^2/ha)					
Ponderosa Pine	**0.007**	0.065	0.141	**0.000**	0.014
Douglas-fir	0.029	**0.008**	0.695	1.00†	0.017
All Species	**0.000**	**0.000**	0.093	**0.000**	0.096
Snags	**0.002**	0.491	0.028†	0.612†	0.713

Bold type indicates significance with a Bonferroni correction of 0.05/5 = 0.01
† indicates cases where a non parametric Wilcoxon signed rank test was used
‡ indicates data was ln transformed

Table 3. Total density summary statistics for ponderosa pine, Douglas-fir, other shade tolerant species and all species pooled. Units are in trees hectare^{-1}.

	Ponderosa pine				Douglas-fir				Other				All Species			
	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K)	Unburn (N)	Logged
Average	145.8	126.4	116.5	267.4	50	179.15	166.52	335.22	1	143	35	52	197	449	318	655
Median	137.5	133.3	110.0	140.0	33.35	133.3	160	150	0	17	0	0	183	408	270	620
25th Percentile	64.6	68.8	70.0	50.0	6.225	83.3	20	20	0	0	0	0	72.95	272.9	200	410
75th Percentile	239.6	175.0	140.0	420.0	93.75	285.425	230	560	2.1	410.4	20	90	308.35	639.6	390	850
IQR	175.0	106.3	70.0	370.0	87.5	202.1	210.0	540.0	2.1	410.4	20.0	90.0	235.4	366.7	190.0	440.0
Minimum-Maximum	8-283	50-200	20-290	0-1,200	0-150	83-392	0-680	0-1,210	0-8	0-417	0-280	0-320	17-433	242-858	40-910	0-1600
Std. Dev.	99.0	58.1	66.6	322.2	55.78	120.65	158.79	363.74	3.39	209.13	70.12	97.19	144.57	231.02	190.69	423.28

Table 4. Detailed size class summary statistics for a) ponderosa pine and b) Douglas-fir. Units are in trees hectare^{-1}. For unburned sites, (K) and (N) refer to sites from Keeling et al (2006) and Naficy (Chapter 1), respectively.

Ponderosa Pine	5-20 cm				20-40 cm				40-60 cm				> 60 cm			
	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K	Unburn (N	Logged	Burned	Unburn (K	Unburn (N	Logged
Average	63.9	38.9	27.0	167.8	33.3	19.4	28.3	72.6	23.6	38.9	30.4	18.7	25.0	29.2	30.9	8.3
Median	37.5	12.5	10.0	60.0	33.3	16.7	10.0	30.0	25.0	25.0	20.0	20.0	25.0	29.2	30.0	10.0
25th Percentile	0.0	8.3	0.0	10.0	6.2	0.0	10.0	10.0	18.8	16.7	10.0	0.0	6.2	20.8	20.0	0.0
75th Percentile	143.7	77.1	40.0	260.0	56.3	37.5	40.0	90.0	33.3	70.9	50.0	30.0	43.8	41.7	40.0	10.0
IQR	143.7	68.8	40.0	250.0	50.0	37.5	30.0	80.0	14.6	54.2	40.0	30.0	37.6	20.9	20.0	10.0
Minimum-Maximum	0-175	8-133	0-120	0-990	0-75	0-50	0-160	0-350	0-33	17-83	0-130	0-90	0-50	Aug-42	0-70	0-40
Std. Dev.	74.3	50.2	35.2	256.3	27.9	19.5	36.6	100.2	12.3	29.2	31.0	24.6	19.7	12.6	19.0	10.3

a)

b)

Douglas-fir	5-20 cm				20-40 cm				40-60 cm				> 60 cm			
	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K)	Unburn (N)	Logged	Burned	Unburn (K	Unburn (N	Logged	Burned	Unburn (K	Unburn (N	Logged
Average	6.9	148.6	108.7	223.5	29.2	22.2	45.7	101.3	9.7	1.4	10.4	8.7	4.2	6.9	1.7	1.7
Median	4.2	95.8	90.0	100.0	12.5	16.7	30.0	60.0	8.3	0.0	0.0	10.0	0.0	4.2	0.0	0.0
25th Percentile	0.0	72.9	10.0	10.0	0.0	6.2	10.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
75th Percentile	16.7	256.225	170	370	58.4	35.4	70.0	240.0	18.8	2.1	20.0	20.0	10.4	12.5	0.0	0.0
IQR	16.7	183.3	160.0	360.0	58.4	29.2	60.0	230.0	18.8	2.1	20.0	20.0	10.4	12.5	0.0	0.0
Minimum-Maximum	0-17	67-325	0-420	0-1,130	0-108	0-67	0-230	0-300	0-25	0-8	0-50	0-40	0-17	0-25	0-10	0-20
Std. Dev.	8.2	106.6	109.6	305.1	42.4	23.4	55.2	107.8	9.7	3.4	14.6	10.6	7.0	9.7	3.9	4.9

Table 5. The a) frequency and b) degree of departure of stand density from reference conditions for unburned and logged sites. In a), values are the percent of sites which depart from the reference interquartile range (IQR) and Range , with % positive and % negative departure in parentheses, for all tree species groups. In b), values represent the average, (median), and minimum-maximum density values for unburned and logged stands which experienced positive departures relative to the upper density limit of the IQR and range of burned stands.

a)

	IQR			Range		
	Unburn (K)	Unburn (N)	Logged	Unburn (K)	Unburn (N)	Logged
All Species	67	47	87	33	17	79
	(67, 0)	(43, 4)	(78, 13)	(33, 0)	(17, 0)	(70, 9)
PIPO	17	31	75	0	4	44
	(0, 17)	(9, 22)	(35, 30)	(0, 0)	(4, 0)	(35, 9)
PSME	67	74	78	33	52	48
	(67, 0)	(61, 13)	(65, 13)	(33, 0)	(52, 0)	(48, 0)
Other	50	39	44	50	39	44
	(50, 0)	(39, 0)	(44, 0)	(50, 0)	(39, 0)	(44, 0)

b)

	IQR			Range		
	Unburn (K)	Unburn (N)	Logged	Unburn (K)	Unburn (N)	Logged
All Species	1.8 (1.6)	1.5 (1.3)	2.6 (2.4)	1.6 (1.6)	1.5 (1.4)	2.0 (1.8)
	1.1 - 2.8	1.0-3.0	1.3-5.2	1.3-2.0	1.1-2.1	1.1-3.7
PIPO	0 (0)	1.1 (1.1)	2.6 (2.0)	0 (0)	1.0 (1.0)	2.2 (1.7)
	0-0	1.1-1.2	1.3-5.0	0-0	1.0-1.0	1.1-4.2
PSME	2.4 (2.0)	2.7 (2.2)	5.4 (4.4)	2.1 (2.1)	1.8 (1.5)	4.0 (3.5)
	1.4-4.2	1.2-7.2	1.2-12.9	1.7-2.6	1.1-4.5	1.0-8.1
Other	143.1 (204.2)	44.4 (40.0)	60.0 (47.5)	35.8 (51.0)	11.1 (10.0)	15.0 (11.9)
	16.7-208.3	5.0-140.0	5.0-160.0	4.2-52.1	1.3-35.0	1.3-40.0

*Values of 1.0 barely exceeded reference IQR or Range. Zero values occur where 100% of sites fell within the reference IQR or Range.

Table 6. Total density (trees/ha) of sites in the ponderosa pine (PIPO) and Douglas-fir/grand fir (PSME) habitat types for logged site density and the density difference between paired logged and unlogged sites.

	Mean	F	p-value
Logged Density			
PIPO	50	9.889	0.005
PSME	746		
Density Difference			
PIPO	-60	6.661	0.017
PSME	397		

Table 7. Regression models of climate and logging variables with a) logged site density (trees/ha) and b) the density (trees/ha) difference (logged-unlogged) between logged and unlogged sites for all small trees < 40 cm DBH, for all Douglas-fir and tree species other than ponderosa pine, and for total stand density.

a)

	Small Trees (<40cm)	Douglas-fir & Other		Total
Model Variables	Logging Intensity	Logging Intensity	Tmax	Logging Intensity
β Coefficient	17.589	10.201	-172.493	7.931
Variable P-value	0.011	0.029	0.107	0.057
Model R-square	0.385	0.187		0.187
Model F-statistic	8.151	3.728		4.13
Model P-value	0.011	0.045		0.057

b)

	Small Trees (<40cm)	Douglas-fir & Other		Total
Model Variables	Logging Intensity	Logging Intensity	Tmax	Logging Intensity
β Coefficient	6.171	8.599	-145.953	5.554
Variable P-value	0.066	0.006	0.041	0.099
Model R-square	0.176	0.426		0.144
Model F-statistic	3.836	6.312		3.022
Model P-value	0.066	0.009		0.099

Table 8. Regional summaries of stand structure for reference sites in ponderosa pine forests of the western U.S. and northern Mexico.

Study Name	Region	Area Name	Forest Age	Historical Conifer Density (trees/ha)	Historical Total Density (trees/ha)	Tree Size (cm)
Keeling et al 2006	Northern Rockies	western MT and central ID	mature/old growth	197 (17-433)	197 (17-433)	44 (26-59)
Arno et al 1995	Northern Rockies	western MT and central ID	mature/old growth	153 (99-212)	153 (99-212)	47 (36-53)
Brown & Cook	Northern Rockies	Black Hills	mixed ages	127 (0-710)	127 (0-710)	51 (0-86)*
Ehle & Baker 2003	Central Rockies (CO)	Rocky Mountain National Park	mature/old growth	144 (68-286)	144 (68-286)	36 (31-41)
Sherriff & Veblen 2006	Central Rockies (CO)	Co Front Range	mixed ages	348-846 (39-1624)	348-846 (39-1624)	--
Harrod et al 1999	Washington	eastern Cascades	mature/old growth	50	50	--
Youngblood et al 2004	Oregon	eastern Cascades	mature/old growth	50	50	60
Fule et al 2002	southwestern U.S.	Grand Canyon	mature	131 (NA)	165 (10-646)	--
Minnich et al 1995	southern CA	San Bernadino Mountains	mostly mature	116 (50-200)	116 (50-200)	most trees > 67
Goforth & Minnich 2008	southern CA	Cuyamaca Mountains	mixed ages	271±82	421±144	--
Stephens & Gill 2005	northern Mexico	Sierra San Pedro Martir	mature/old growth	145 (30-320)	145 (30-320)	33 (10-110)

* Values are quadratic mean diameter, which tend to weight large trees