MONITORING LANDSCAPE-SCALE FOREST STRUCTURE AND POTENTIAL FIRE BEHAVIOR CHANGES FOLLOWING PONDEROSA PINE RESTORATION TREATMENTS

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Abstract

Monitoring Landscape-Scale Forest Structure and Potential Fire Behavior Changes Following Ponderosa Pine Restoration Treatments

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We evaluated landscape-scale forest restoration treatment implementation and effectiveness and assessed canopy fuels and potential fire behavior changes following landscape-scale forest restoration treatments in a ponderosa pine forest at Mt. Trumbull, Arizona. The goal of the project was to alter forest structure by thinning and burning to more closely resemble forest conditions prior to Euro-American settlement in 1870. We measured 117 permanent plots before (1996/97) and after (2003) treatments. The plots were evenly distributed across the landscape and represented an area of approximately 1200 ha, about half of which was an untreated control. The success of treatment *implementation* was variable. Most of the area originally planned for restoration was treated in some manner by 2003; however, only 70% received the full planned treatment (thin and burn). Although pine density decreased significantly in the treated area, the projected residual density was exceeded by 111-256%. Despite contract amendments to terminate oak cutting, some oaks were still cut for several reasons. Thirteen percent of the presettlement pines died in the treated area by 2003 slightly exceeding the 10% maximum allowable mortality outlined by managers; however, 9% percent of the presettlement pines died in the control. One-third of large snags were lost, falling below

the snag retention target, but new large snags were recruited, resulting in a net increase in snag density within the treated area. Implementation goals for large logs were achieved. Restoration treatments decreased canopy fuel load (CFL) and canopy bulk density (CBD) in the treated area, while slight increases occurred in the control. Predicted outcomes were consistent between the two fire behavior models (FlamMap and Nexus): under extreme drought and wind conditions, active crown fire hazard was reduced in the treated area. In contrast, the models show little change in active crown fire hazard in the control over the same time period. Although restoration treatments were not implemented perfectly, they were *effective* in attaining the overall project goal of restoring more open forest structure conditions while preserving most of the presettlement trees. Furthermore, canopy fuels and active crown fire hazard were substantially reduced, allowing for the reintroduction of low-intensity surface fires.

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Preface

This thesis is divided into five chapters. Because it was written in manuscript format, some redundancy occurs throughout the thesis. Chapter 1 is an overall introduction and describes the Mt. Trumbull restoration project and the two studies within the thesis. Chapter 2 is a literature review associated with this study and includes discussion of ecological restoration of ponderosa pine, adaptive management, and fire behavior modeling. Chapter 3 and Chapter 4 were written in manuscript format for submission to peer-reviewed scientific journals. Chapter 5 summarizes the main findings of this study and discusses the associated management implications and future research.

Chapter 1

Introduction

Southwestern ponderosa pine ecosystems have undergone substantial changes in forest structure, fuel loads, and crown fire hazard due to the cessation of the historical fire regime (Cooper 1960, Covington and Moore 1994*a*, 1994*b*). Prior to European settlement, ponderosa pine (*Pinus ponderosa* P. & C. Lawson var. *scopulorum* Engelm.) forests were historically comprised of a matrix of native herbaceous species interspersed with patches of large, mature trees (White 1985, Covington et al. 1997) and a smaller component of seedlings and saplings (Mast et al. 1999, Bailey and Covington 2002). This open, park-like structure was maintained by frequent, surface fires until fire exclusion occurred in the late 1800's associated with Euro-American settlement (Covington and Moore 1994*a*). By the mid-1900's, foresters recognized that excessive livestock grazing, logging, and fire suppression had contributed to increased tree density and crown fire hazard (Pearson 1910, Weaver 1951).

In the mid-1990's, the Bureau of Land Management (BLM) and the Ecological Restoration Institute (ERI) at Northern Arizona University (NAU) initiated a landscape-scale restoration project (~1500 ha) intended to restore ecosystem health and reduce crown fire hazard while monitoring a wide variety of ecosystem components and processes in order to adapt restoration practices based on observation of treatment effects (Moore et al. 2003). Prior to treatment, the ERI installed a grid of permanent plots across the project area (including a ~500 ha untreated control) to collect data on contemporary and historical forest conditions. In addition, fire scars were collected throughout the landscape and indicated that the long-term frequent fire regime was abruptly interrupted in 1870. Restoration treatments were implemented beginning in 1996 and included a thinning prescription designed to emulate presettlement forest structure conditions followed by prescribed surface fire. In 2003, a subset of plots was remeasured representing a large portion of the project area (~700 ha treated area, ~500 ha control).

Fire behavior models are important tools for fire managers and are often used to develop and evaluate alternative treatments, to estimate fire effects, to assess risk to life and property, and to understand ecosystems (Andrews and Queen 2001, Reinhardt et al. 2001). In this study, we used two models to assess the effectiveness of restoration treatments on crown fire hazard at Mt. Trumbull. FlamMap (Finney, in preparation), a GIS-based system, assesses fuel hazards and uses terrain, fuels, and weather inputs to predict potential fire behavior for each individual pixel on the raster landscape simultaneously. Nexus, another hazard model, uses plot- or stand-level data to predict potential fire behavior (Scott and Reinhardt 1999, 2001).

Canopy fuels are a crucial input for both models but they are rarely measured directly. Brown (1978) provided allometric equations for ponderosa pine that have been widely applied. In Arizona, Fulé et al. (2001, 2004) applied locally developed allometric equations that predicted less canopy fuel, and hence lower canopy bulk density, than would have been predicted by Brown's (1978) equations. Cruz et al. (2003) developed stand-level equations to predict canopy fuels based on tree density and basal area. Because these three approaches differ, the selection of a canopy fuel modeling approach may affect fire behavior model results. We compared canopy fuel estimates produced by all three equations and used each as an input for both FlamMap and Nexus.

The goal of this study was to provide feedback from monitoring for the adaptive management process in the Mt. Trumbull restoration project. Although several smaller adaptive changes had already occurred throughout the course of the project, this study represents the first landscape-scale evaluation of treatment implementation and effectiveness based on forest structure monitoring data collected before and after treatment. The objectives of this study were to: 1) determine whether restoration treatments were implemented as planned; 2) determine whether the treatments were effective in terms of the ultimate ecological restoration goals; 3) compare three common canopy fuel estimation approaches using data collected in this study; 4) compare the output from FlamMap and Nexus; and 5) assess the effectiveness of landscape-scale restoration treatments on reducing crown fire hazard.

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Chapter 2

Literature Review

Introduction

The degradation of Southwestern ecological systems and the associated threat to biodiversity was addressed by Aldo Leopold (1924, 1934) in the early 20th century. Today, concerns about increasing stand-replacing crown fires, insect outbreaks, and pathogen epidemics in western United States forested ecosystems have brought the topic to the environmental, economic, and political forefront (Covington 2000, Covington 2002). According to the U.S. General Accounting Office, “the number of uncontrollable and catastrophically destructive wildfires is the most extensive and serious national forest health-related problem in the interior West” (U.S GAO 1998). Presettlement ponderosa pine (*Pinus ponderosa* P. & C. Lawson var. *scopulorum* Engelm.) forests were historically comprised of a matrix of native shrub, grass, and forb species interspersed with patches of large, mature trees (White 1985, Covington et al. 1997) and a smaller component of seedlings and saplings (Mast et al. 1999, Bailey and Covington 2002). This open, park-like structure was maintained by frequent, low-intensity surface fires until fire exclusion occurred in the late 1800’s associated with Euro-American settlement (Covington and Moore 1994*a*). Ponderosa pine ecosystems throughout the southwestern United States have become uncharacteristically dense and structurally homogeneous as a result of heavy livestock grazing, logging, fire exclusion, and climatic factors (Cooper 1960, Covington and Moore 1994*b*, Covington et al. 1997). These changes in fire regime and forest structure have increased vulnerability of ponderosa pine forests to large, stand-replacing crown fires that endanger human and ecological communities (Allen et al. 2002). Ecological restoration provides an opportunity to restore the natural structure and function of these fire-adapted ecosystems consistent with their evolutionary environment (Moore et al. 1999).

Ecological Restoration of Ponderosa Pine Ecosystems

The first attempts at restoring ponderosa pine forest ecosystem structure and function began with the reintroduction of surface fire (Weaver 1951, Biswell 1972, Covington and Sackett 1984). Prescribed fires reduced fuel loading and increased forest floor nutrients (Covington and Sackett 1992), but did not reduce continuous vertical fuels in the form of small diameter trees nor prevent mortality in presettlement trees (Sackett and Hasse 1998). In the early 1990's an intensive, fine-scale restoration experiment at Gus Pearson Natural Area (GPNA) combined mechanical thinning of small-diameter trees with prescribed surface fire (Covington et al. 1997). This resulted in forest structure closer to the desired presettlement conditions as well as improved ecosystem function (Covington et al. 1997, Kaye et al. 2005) and improved health and vigor of older/larger trees (Kolb et al. 1998, Feeney et al. 1998, Stone et al. 1999, Skov et al. 2004). However, it was not possible to repeat the intensive restoration treatments implemented at GPNA on a landscape-scale or to extrapolate GPNA findings to larger tracts of land because of the unique characteristics of the never-logged Natural Area. Therefore, landscape-scale restoration research was necessary for further understanding of southwestern ponderosa pine ecosystems.

Mt. Trumbull

Mt. Trumbull is located in the Uinkaret Mountains on the Arizona Strip in the Grand Canyon-Parashant National Monument and managed by the Bureau of Land Management (BLM). A 7,000 ha ponderosa pine ecosystem lies in a broad saddle between Mt. Trumbull and Mt. Logan, two extinct volcanoes (Friederici 2003). Native Americans, who named the mountains the Uinkarets or "region of pines," inhabited the Mt. Trumbull area for millennia before it was settled by Euro-Americans circa 1870 (Altschul and Fairley 1989). Records from fire-scarred trees suggest that relatively open forest structure was maintained by a frequent fire regime prior to Euro-American settlement in 1870 (Waltz and Fulé 1998, Heinlein et al. 1999). Shortly thereafter, a small sawmill was constructed at Nixon Springs and logging began mainly for the construction of the St. George Mormon Temple (R. Davis, pers. comm.). The mill could not process trees greater than 32" dbh (81 cm), therefore the largest ponderosa pines were not harvested and many remain throughout the Mt. Trumbull area today (Moore et al. 2003). Livestock grazing removed the fine fuels that supported frequent, low-intensity fires prior to settlement and most ignitions that occurred during the latter 20th century were suppressed (Altschul and Fairley 1989). These land use changes facilitated ponderosa pine seedling irruptions outside the historic range of variability consistent with those documented throughout the southwestern United States (Cooper 1960, White 1985, Savage et al. 1996).

Initiated in 1995, the Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project is the longest running landscape-scale ponderosa pine restoration project in the Southwest (Friederici 2003) and the first to incorporate operational treatments and intensive monitoring. The primary goal of the project was to restore forest structure and ecosystem processes to within the historical range of variability, as well as reduce fuel loads, disrupt fuel continuity, and reduce the likelihood of stand-replacing crown fires by implementing landscape-scale mechanical thinning followed by prescribed surface fire (Moore et al. 1999, Moore et al. 2003). Another goal of the project was to monitor a wide variety of ecosystem components and processes in order to adapt restoration practices based on observation of treatment effects (Moore et al. 2003).

Adaptive Management and Monitoring

Adaptive management (AM) is the process of adjusting management actions based on monitoring information (Holling 1978, Walters 1986). This "learn by doing" approach has been used in natural resource management since the 1970's; it involves setting goals, planning, and commitment to monitoring. AM is classified into two types: "passive" and "active" (Walters and Holling 1990). In passive AM, the management action considered best is designed and implemented and adjustments are made based on monitoring and evaluation. In active AM, a range of alternatives are designed and implemented, then monitoring and evaluation are used to identify which alternative was most effective in meeting objectives, and finally adjustments to subsequent management decisions are made based on those conclusions (Murray and Marmorek 2003).

Monitoring, or repeated measurement of ecological variables through time, is crucial for evaluating management actions (Fulé 2003). *Implementation* and *effectiveness* monitoring are particularly important when adaptive management principles are used (Block et al. 2001, Moir and Block 2001). Implementation monitoring is defined as "the process of determining if a planned activity was accomplished" (Noss and Cooperrider 1994). Implementation monitoring reveals to managers whether the treatments were implemented as originally prescribed. It would be difficult to justify changing treatment prescriptions if managers were unsure whether the original prescriptions were followed. Effectiveness monitoring is defined as "the process of determining if an activity achieved the stated goal or objectives" (Noss and Cooperrider 1994), and allows managers to determine if objectives were met and whether or not to alter implementation methods or treatments. The implementation of treatments and the ultimate effectiveness of treatments in a project may not necessarily have the same success. For example, the goals of a project could be met even if the treatments were not implemented as planned. Alternatively, the treatment could be carried out precisely, but the desired goals may not be met at all.

Reference Conditions

Ecological restoration often aims to repair degradation by re-establishing the historical composition, structure, and function of indigenous ecosystems (Society for Ecological Restoration 1993). Reference conditions refer to the historical range of variability of ecosystem structures and processes (Morgan et al. 1994, Landres et al. 1999) and are often used as a baseline to evaluate restoration treatment effectiveness (Moore et al. 1999). In southwestern ponderosa pine, reference conditions are usually determined based on the date of Euro-American settlement which often coincides with the date of the last widespread surface fire; however, historical records, plot data, photographs and accounts can also be used as supplemental lines of evidence (Moore et al. 1999). Reference conditions can be further corroborated with contemporary reference sites where historical forest structure and fire regimes are relatively intact (Fulé and Covington 1996, Fulé and Covington 1999, Fulé et al. 2002*a*, Fulé et al. 2003, Stephens and Fulé 2005). Nonetheless, many authors argue that using ecosystem conditions present at the time of Euro-American settlement as a point of reference is subjective and arbitrary and should not be used exclusively as a guide for future management (Southwest Forest Alliance 2000, Wagner et al. 2000).

Types of Wildland Fire

Wildland fire is classified into ground (sub-surface), surface, or crown categories based on where in the fuel strata burning occurs (Pyne et al. 1996). Crown fires are further subdivided into three types: passive, active, and independent (Van Wagner 1977). Passive crown fire, or torching, occurs when fire transitions from the surface and ignites the lower canopy. The windspeed at which torching is initiated, the "torching index," is largely a function of canopy base height. Active crown fires burn the entire surface/canopy fuel complex, depending primarily on the bulk density of foliage and fine twigs in the canopy. Independent crown fires, or active crown fires that do not rely on surface fire, are extremely rare and not considered further here. Since passive and active crown fire behavior are linked to different canopy fuel variables, it is possible to encounter a situation where passive crown fire is not predicted to occur, due to a high canopy base height, but active crown fire could occur, due to high canopy bulk density. Scott and Reinhardt (2001) described this hysteresis as a "conditional" surface fire; active crown fire could occur on the condition that canopy burning entered the stand from outside, otherwise surface fire would occur.

Fuel Treatment Effects on Fire Behavior

The fire environment triangle is comprised of three influencing forces: fuel, weather, and topography (Pyne et al. 1996). The fuels leg is most related to forest structure and is the only one that can be altered by management actions (Agee 1996). Many restoration and fuel treatment projects have been implemented throughout the western United States to restore natural ecosystem structure and function and to reduce the threat of stand-replacing crown fires (Scott 1998, Lynch et al. 2000, Fulé et al. 2001*a*, Stratton 2004). Such studies that examine treatment effects on fire behavior or severity can be classified into three categories: experimental, observational, and modeling. Experimental studies test fire behavior by purposely igniting fires and examining the effects during and after the burn. Although researchers have deliberately ignited crown fires to study their properties in certain isolated settings (Alexander et al. 2004), most experimental studies are focused on effects of relatively low-intensity fires (Weaver 1957, Covington et al. 1997, Fulé et al. 2002*b*) because intentionally lighting large, stand-replacing crown fires is difficult to justify. Observational studies examine the effects of wildfires after they occur. Pollet and Omi (2002), Martinson and Omi (2003), Graham (2003), and Cram and Baker (2003) examined fire effects in treated and untreated forest stands in several western states, finding that treated stands generally show lower fire severity, although treatments did not necessarily preclude severe burning or prevent the passage of landscape-scale crown fires. Following the 2002 Rodeo-Chediski fire in

Arizona, Finney et al. (2005) used satellite imagery and Strom (2005) used ground data to show that burning and/or cutting + burning treatments substantially reduced fire severity. Observational approaches are essential for measuring real-world effects of treatments, but the scope of inference of the approach remains limited by lack of pre-fire data, randomization, and replication. The final technique, fire behavior modeling, is the most removed from actual fire behavior but the most flexible for testing alternative scenarios of stand development, treatments, or weather conditions. Various studies have used models to evaluate potential fire behavior after restoration or fuel treatments at scales ranging from stands (Stephens 1998, Fulé et al. 2001*a* & 2001*b*, Fulé et al. 2002*b*,) to landscapes (Fiedler and Keegan 2003, Fulé et al. 2004, Stratton 2004).

Fire Behavior Modeling

Fire behavior models are important tools for fire managers and are often used to develop and evaluate alternative treatments, to estimate fire effects, to assess risk to life and property, and to understand ecosystems (Andrews and Queen 2001, Reinhardt et al. 2001). Deterministic semi-empirical fire behavior models based on Rothermel's (1972) surface fire model coupled with canopy initiation and spread models are widely used in fire behavior analysis. FARSITE, a GIS-based system, uses terrain, fuels, and weather inputs to simulate the growth, spread, and behavior of wildland fires (Finney 1998). The variant FlamMap (Finney, in preparation), adapted for assessing fuel hazards, uses most of the same inputs as FARSITE but predicts potential fire behavior for each individual pixel on the raster landscape simultaneously. Nexus, another hazard model, uses plot- or stand-level data to predict potential fire behavior (Scott and Reinhardt 1999, 2001).

FlamMap and Nexus differ in crown fire outputs provided, with FARSITE simulating only passive and active crown fire, while Nexus also provides estimates of conditional surface fire.

Canopy Fuel Estimation and Measurement

Canopy fuels are a crucial input for models that predict crown fire but they are rarely measured directly. Brown (1978) provided allometric equations for ponderosa pine that have been widely applied. In Arizona, Fulé et al. (2001*a*, 2004) applied locally developed allometric equations that predicted less canopy fuel, and hence lower canopy bulk density, than would have been predicted by Brown’s (1978) equations. Cruz et al. (2003) developed stand-level equations to predict canopy fuels based on tree density and basal area. Because these three approaches differ, the selection of a canopy fuel modeling approach may affect fire behavior model results.

Conclusion

Implementation of ecological restoration and other fuel treatments in southwestern ponderosa pine forests will likely increase in the future. It is also likely that the scale of these projects will broaden from stand- and landscape-scale to the scale of entire watersheds. It will be important that long-term monitoring and research also focus on broader scales so that we can avoid exclusively extrapolating information from fine to broad scales.

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Chapter 3

Monitoring Landscape-Scale Forest Structure Changes Following Ponderosa Pine Restoration Treatments

Abstract

We evaluated landscape-scale forest restoration treatment implementation and effectiveness in a ponderosa pine forest at Mt. Trumbull, Arizona. The goal of the project was to alter forest structure by thinning and burning to more closely resemble forest conditions prior to Euro-American settlement in 1870. We measured 117 permanent plots before (1996/97) and after (2003) treatments. The plots were evenly distributed across the landscape and represented an area of approximately 1200 ha, about half of which was an untreated control. For tree density, we evaluated implementation success based on the projected density of ponderosa pine; we evaluated treatment implementation and effectiveness for other variables based on goals outlined by managers or 1870 conditions. The success of treatment implementation was variable. About 94% of the area originally planned for restoration was treated in some manner by 2003; however, only 70% received the full planned treatment (thin and burn). Although pine density was reduced significantly by 66% from 428.6 pines/ha to 146.3 pines/ha in the treated area, the projected density was exceeded by 111-256%. Pine density exceeded the projected outcome by only 10-85% when only thinned and burned plots were analyzed. Despite contract amendments to terminate oak cutting, 32.3 oaks/ha were cut even after the decision to stop. Thirteen percent of the presettlement pines died in the treated area by 2003, slightly exceeding the 10% maximum allowable mortality outlined by managers;

however, 9% percent of presettlement pines also died in the control, indicating that presettlement pines in untreated areas were nearly as vulnerable as those exposed to restoration treatments. Goals for large snag retention were not achieved but new snags were recruited above expectations, resulting in a 45% net increase in snag density within the treated area. Sixty-five percent of logs >50 cm were retained, thus, implementation goals for large logs were achieved. Although restoration treatments were not implemented perfectly, they were *effective* in attaining the overall project goal of restoring more open forest structure conditions while preserving more than 75% of the presettlement pines. Furthermore, canopy fuel loads were substantially reduced, allowing for the reintroduction of low-intensity surface fires. The Mt. Trumbull restoration project serves as a useful example of a collaborative effort between managers and researchers with a strong commitment to monitoring ecological responses to restoration treatments.

Key Words: adaptive management, ecological restoration, forest structure, landscape-scale, monitoring, Mt. Trumbull, ponderosa pine

Introduction

Changes in fire regime and forest structure have increased vulnerability of ponderosa pine forests to large, stand-replacing crown fires that endanger human and ecological communities (Allen et al. 2002). Ponderosa pine (*Pinus ponderosa* P. & C. Lawson var. *scopulorum* Engelm.) ecosystems throughout the southwestern United States have become uncharacteristically dense and structurally homogeneous as a result of heavy livestock grazing, logging, fire exclusion, and climatic factors (Cooper 1960, Covington and Moore 1994*a*, 1994*b*, Covington et al. 1997). Ecological restoration provides an opportunity to restore the natural structure and function of these fire-adapted ecosystems consistent with their evolutionary environment (Moore et al. 1999). Landscape-scale monitoring is necessary to understand how restoration practices affect sizeable tracts of land as restoration implementation becomes more prevalent.

Initiated in 1995, the Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project is the longest running landscape-scale ponderosa pine restoration project in the Southwest (Friederici 2003) and the first to incorporate operational treatments and intensive monitoring. The primary goal of the project was to restore forest structure and ecosystem processes within the historical range of variability, as well as reduce fuel loads, disrupt fuel continuity, and reduce the likelihood of stand-replacing crown fires by implementing landscape-scale mechanical thinning followed by prescribed surface fire (Moore et al. 1999, Moore et al. 2003). Finally, we aimed to monitor a wide variety of ecosystem components and processes in order to adapt restoration practices based on observations of treatment effects.

Adaptive management (AM) includes the process of adjusting management actions based on monitoring information (Holling 1978, Walters 1986). This "learn by doing" approach has been used in natural resource management since the 1970's; it involves setting goals, planning, and commitment to monitoring. AM is classified into two types: "passive" and "active" (Walters and Holling 1990). In passive AM, the management action considered best is designed and implemented and adjustments are made based on monitoring and evaluation. In active AM, a range of alternatives are designed and implemented, then monitoring and evaluation are used to identify which alternative was most effective in meeting objectives, and finally adjustments to subsequent management decisions are made based on those conclusions (Murray and Marmorek 2003).

Monitoring, or repeated measurement of ecological variables through time, is crucial for evaluating management actions (Fulé 2003). *Implementation* and *effectiveness* monitoring are particularly important when adaptive management principles are used (Block et al. 2001, Moir and Block 2001). Implementation monitoring is defined as “the process of determining if a planned activity was accomplished” (Noss and Cooperrider 1994). Implementation monitoring reveals to managers whether the treatments were implemented as originally prescribed. It would be difficult to justify changing treatment prescriptions if mangers were unsure whether the original prescriptions were followed. Effectiveness monitoring is defined as “the process of determining if an activity achieved the stated goal or objectives” (Noss and Cooperrider 1994), and allows managers to determine if objectives were met and whether or not to alter implementation methods or treatments. The implementation of treatments and the ultimate effectiveness of treatments in a project may not necessarily have the same success. For example, the goals of a project could be met even if the treatments were not implemented as planned. Alternatively, the treatment could be carried out precisely, but the desired goals may not be met at all.

Our objective in this study was to evaluate monitoring data and apply it to the adaptive management process in the Mt. Trumbull restoration project. Although several smaller adaptive changes had already occurred throughout the course of project implementation (Waltz et al. 2000), this study represents the first landscape-scale evaluation of treatment implementation and effectiveness based on forest structure monitoring data collected before and after treatment. Specifically, we intend to answer two overarching questions: 1) Were restoration treatments implemented as planned?, and 2) Were the treatments effective in terms of the ultimate ecological restoration goals?

Methods

Study Area

Mt. Trumbull is located in the Uinkaret Mountains on the Arizona Strip in the Grand Canyon-Parashant National Monument and managed by the Bureau of Land Management (BLM). Vegetation in the study area (elevation 2,000 to 2,250 m) is comprised of ponderosa pine and Gambel oak (*Quercus gambelii* Nutt.) with Utah juniper (*Juniperus osteosperma* [Torr.] Little), pinyon pine (*Pinus edulis* Engelm.), New Mexico locust (*Robinia neomexicana* Gray) and several shrub species occurring throughout the area. Soils are derived from basaltic parent material. The two main soil types found in the study area are the Wutoma-Lozinta complex which consists of ashy-skeletal over fragmental or cindery, mixed, mesic Vitrandic Haplustepts, and Sponiker soils, classified as fine, smectitic, mesic Pachic Argiustolls (Natural Resources Conservation Service 2004).

Native Americans, who named the mountains the Uinkarets or "region of pines," inhabited the Mt. Trumbull area for millennia before it was settled by Euro-Americans circa 1870 (Altschul and Fairley 1989). Records from fire-scarred trees suggest that relatively open forest structure conditions were maintained by a frequent fire regime prior to Euro-American settlement in 1870 (Waltz and Fulé 1998, Heinlein et al. 1999, Fulé, unpublished data).

Annual precipitation at Nixon Flats (elevation 1,981 m, approximately 3 km NE of study site) averaged 47.2 cm with an average January temperature of 1°C and an average July temperature of 21°C between January 1992 and December 2003 (Western Regional Climate Center 2005). Annual precipitation at Mt. Logan (elevation 2,195 m, approximately 2 km SW of study site) averaged 31.2 cm with an average January temperature of -1°C and an average July temperature of 20°C between January 1986 and December 2003 (Western Regional Climate Center 2005). Most precipitation occurs in winter and during summer monsoon storms; spring and fall are relatively dry.

About half of the approximately 1200 hectare study landscape (Figure 3.1) is a contiguous, "untreated", densely-treed area (hereafter "control area" or "control"). The other half, hereafter "treated area", is adjacent to the control. Restoration treatments were carried out between 1996 and 2003 (Moore et al. 2003). Some untreated areas remain within the treated area boundary, such as controls for other experiments, or operationally inaccessible areas, ranging from approximately 10 to 40 hectares.

Restoration Treatment Prescriptions

The thinning design was based on the presettlement (pre-1870) pattern of tree species composition and spatial arrangement (Covington et al. 1997, Waltz et al. 2003). All living ponderosa pines older than 1870 or larger than 70 cm dbh were retained (Moore et al. 2003); presettlement ponderosa pines of any size were identified in the field based on yellow bark coloration and tree characteristics (White 1985). In addition, wherever evidence of presettlement remnant ponderosa pine material was encountered (i.e., snags, logs, stumps, stump holes), 1.5 postsettlement ponderosa pine replacement trees (if >40.6 cm diameter) or 3 ponderosa pine replacement trees (if ≤40.6 cm diameter) were retained within an approximately 18.2 m search radius. An implementation objective was to retain the presettlement ponderosa pines that were still alive, plus leave up to 300% more ponderosa pine trees than were present prior to 1870. The surplus of retained trees was intended to account for the smaller biomass contributed by smaller diameter replacement trees, possible loss of presettlement evidence, and to allow a margin for unintended mortality due to restoration treatments (Covington et al. 1997). Because postsettlement replacement trees were located near remnant evidence of presettlement structures, the spatial variability that existed prior to disturbance of the historical fire regime was reflected in the post-treatment forest structure. Therefore, rather than a "one-size-fits-all" approach, areas that were relatively open in 1870 (i.e., few remnants found) would be relatively open after treatment and areas that were relatively dense in 1870 would be relatively dense after treatment. Originally, Gambel oak trees were also thinned, but given high oak mortality due to prescribed burning, these guidelines were modified early in the project to terminate oak cutting. All living trees of other species (e.g., New Mexico locust) were not cut because they were so few in number. Unmarked trees were commercially logged or non-commercially thinned in this leave-tree thinning. Slash was lopped and scattered and was crushed by a bulldozer in some areas (Jerman et al. 2004). Prescribed burn preparation included raking accumulated forest floor material away from living presettlement trees to prevent cambial girdling (Sackett et al. 1996) and from large snags to limit ignition (Moore et al. 2003).

Prescribed fires were often ignited at night when humidity was relatively high. It is important to note that although most of the treatments were completed by the time of our measurement in 2003, there were portions that were thinned only or burned only.

Field Methods

Prior to treatment in 1996 and 1997, we installed 117 permanent plots on a 300 meter grid (Figure 3.1) throughout the Mt. Trumbull landscape as part of a before-after-control-impact (BACI) study design (Stewart-Oaten and Bence 2001); all plots (55 control, 61 treated, 1 partially treated, excluded from analysis) were remeasured in the summer of 2003. The plots were adapted from the National Park Service's Fire Monitoring plots (Reeberg 1995, NPS 2003), with modifications to collect detailed tree condition and dendroecological data for reconstruction of historical forest structure. Sampling plots were 0.1 ha (20 x 50 m) in size, oriented with the 50-m sides uphill-downhill to maximize sampling of variability along the elevational gradient and to permit correction of the plot area for slope.

Overstory trees, those larger than 15 cm diameter at breast height (dbh) were measured on the entire plot (1000 m^2) and trees between 2.5-15 cm dbh (pole-sized trees) were measured on one quarter-plot (250 m^2); all trees were tagged. Tree attributes measured were: species, dbh, and condition class [(1) live; (2) declining; (3) recent snag; (4) loose bark snag; (5) clean snag; (6) snag broken above breast height; (7) snag broken below breast height; (8) downed dead tree; (9) stump (Thomas et al. 1979); and (10), stump hole]. Total height was measured for pole-sized trees but not for overstory trees during the pre-treatment measurement; total height and crown base height were measured for all trees in 2003. All overstory and pole-sized trees were also mapped within the larger 1000 m^2 plot. Regeneration (seedlings and saplings <2.5 cm dbh) was tallied by species, condition, and height class in a 50 m^2 subplot. Ponderosa pine trees were considered potentially presettlement if dbh ≥37.5 cm or if bark was yellowed (White 1985). Trees of all other species were considered potentially presettlement if dbh ≥17 cm dbh (Barger and Ffolliott 1972). Tree cores were collected at 40 cm above ground level for all potentially presettlement trees and for a random 10% subsample of all other live trees ≥2.5 cm to determine past size, as described below. Canopy cover measured by vertical projection (Ganey and Block 1994) was recorded at 3 m intervals along the two 50-m sidelines of each plot for a total of 32 points per plot. Post-treatment measurements on plots coincided as closely as possible to the original day and month of the original measurement.

Reconstruction Methods

Tree increment cores were surfaced and crossdated (Stokes and Smiley 1968) using locally developed tree-ring chronologies. Rings were counted on cores that could not be crossdated, especially young trees and junipers. Additional years to the center were estimated using a pith locator (concentric circles matched to the curvature and density of the inner rings) for cores without a pith (Applequist 1958).

We reconstructed forest structure using dendroecological methods described in detail by Fulé et al. (1997) and Mast et al. (1999). We reconstructed diameter for all living trees by subtracting the radial growth since 1870 measured on increment cores and estimated death date of dead trees based on tree condition class using diameter dependent snag decomposition rates (Thomas et al. 1979, Rogers et al. 1984). We performed a sensitivity analysis by using the 25^{th}, 50^{th}, and 75^{th} percentile decomposition rates to examine the effect of slower or faster decomposition on estimates of death date and 1870 structure. Less than ±1% change in reconstructed forest structure occurred during this analysis, so the 50^{th} percentile reconstruction was used in this study.

Forest structure reconstruction methods were based on the assumption that evidence of all structures (i.e., snags, logs, stumps, stump holes) present in 1870 was intact, located, and correctly identified during the pre-treatment inventory. The probability that this occurred was relatively high given the absence of fire combined with the semi-arid environment limiting the decomposition of conifer wood (Fulé et al. 1997, Mast et al. 1999, Waltz et al. 2003), and because field crews were trained to identify the presence and species of presettlement structures. Moore et al. (2004) found that reconstruction field techniques in a similar environment and forest type were reliable within ±10% of tree density over ~90 years.

Evaluation of Treatment Implementation and Effectiveness

We compared the projected outcomes to actual outcomes of tree density to evaluate whether or not thinning prescriptions were implemented as planned. Because the thinning prescriptions were based on living and dead presettlement ponderosa pine evidence that we measured on the plots, we evaluated the success of implementation quantitatively based on the projected outcome for ponderosa pine tree density calculated as the sum of all living presettlement trees plus 150-300% of the presettlement remnant evidence (1.5 - 3 postsettlement trees retained per presettlement evidence). The prescription called for retaining the living trees of the other species (with exception of some oak cutting early on in the project) based on the assumption that prescribed burning would thin these trees. Because a projected tree density was not set for non-ponderosa pines, we simply reported on cutting and mortality for these species. We evaluated treatment implementation and/or effectiveness for presettlement tree cutting and mortality, and snag and log densities based on goals outlined by managers and used 1870 conditions to evaluate diameter distributions, regeneration, and canopy cover.

Statistical Analysis

We tested whether the control and treated areas differed in live tree density and basal area, snag and log density, regeneration density, and canopy cover before (1996/97) and after treatment (2003). Univariate analyses examined total response (e.g., total live tree density across all species). Multivariate analyses examined the composition of the total response (e.g., matrix of live tree density of each species).

Univariate analyses were made using Wilcoxon tests to obtain a *Z*-score for each test in 1996/97 and in 2003. Changes in tree density and basal area from 1870 to 1996/97 across the entire landscape were analyzed with Wilcoxon signed ranks tests (T^{+}) because the control and treated areas were combined (Conover 1999). The alpha chosen for all analyses was 0.05.

Multivariate analyses were made using a permutation-based ANOVA with DISTLM software (Version 5.0; Anderson 2005). This procedure permits the analysis of univariate or multivariate data using any distance measure and linear model. The calculated statistic is termed a ‘pseudo-*F*’ and is calculated, like a traditional *F*-statistic, as the sum of the squared distances among groups divided by the sum of the squared distances within groups (see Anderson (2001) and McArdle and Anderson (2001) for details). Data were untransformed and unstandardized. Dissimilarity matrices were calculated using the Bray-Curtis distance measure as this distance measure is appropriate for most ecological data (Faith et al. 1987). *P*-values were calculated by permuting the observations 9999 times, so no assumptions of the distributional form of the data were required.

Results

Reconstructed tree density and basal area indicated that relatively open conditions (average 97.3 trees/ha and 9.5 m²/ha of basal area) existed over the entire landscape in 1870. Ponderosa pine made up 95% of the total basal area but only 38% of total tree density (Table 3.1). The control and treated areas did not differ in terms of total tree density (Z=-0.9, P=0.367) in 1870; however, total basal area was slightly greater (Z=2.0, P=0.041) in the control (Table 3.1). The treated and control areas did not differ in terms of tree density composition (pseudo-F=2.6, P=0.055) or basal area composition (pseudo-F=2.5, P=0.052) in 1870. By 1996/97, total tree density (T^+=3392, P<0.001) and total basal area (T^+=3325, P<0.001) increased significantly (779% increase and 245% increase, respectively) since 1870 across the entire landscape. While total tree density (Z=0.5, P=0.584), total basal area (Z=-0.4, P=0.676), and basal area composition (pseudo-F=0.04, P=0.952) in 1996/97 did not differ between treated and control areas, tree density composition (pseudo-F=2.8, P=0.033) was significantly different between the treated area and control. Ponderosa pine dominated the 1996/97 pre-treatment landscape making up 73% of the total trees/ha and 86% of total basal area in the control area and comprising 55% of the total trees/ha and 79% of the total basal area in the treated area.

By 2003, total tree density (Z=5.0, P<0.0001) and total basal area (Z=5.1, P<0.001) were significantly decreased in the treated area (Table 3.1) and tree density composition (pseudo-F=17.1, P=0.0001) and basal area composition (pseudo-F=12.9, P=0.0001) were significantly different between the treated area and control. Although post-treatment forest structure was relatively open in the treated area, there were a range of densities ranging from extremely open sites (minimum of 10 trees/ha) to sites that were relatively dense (maximum of 1581 trees/ha). There were also relatively open areas in the control (minimum of 108 trees/ha), but the most dense areas (maximum of 3337 trees/ha) were more than twice as dense as the most dense areas in the treated area. The variability throughout the landscape in all time periods is evident when stand density index (SDI), a measure of density that incorporates trees/ha and basal area/ha (Reineke 1933), is interpolated across the plot grid (Figure 3.2). In 2003, ponderosa pine comprised 72% of total basal area but only made up 37% of the total trees/ha in the treated area. Conversely, Gambel oak made up 56% of the total trees/ha but only accounted for 26% of the basal area. The other three species were relatively sparse throughout the landscape in all time periods. Tree density and basal area were even lower in the 2003 treated area when plots that were thinned only, burned only, or not treated were excluded from analysis. Sites that were thinned and burned had 73% fewer trees/ha and 55% less basal area/ha compared to pre-treatment levels (Table 3.2).

Nearly two-thirds of the overall reduction in total tree density and three-quarters in total basal area between 1996/97 and 2003 in the treated area was due to the thinning of ponderosa pine. The thinning of other species accounted for only about 3% of the overall decrease in total tree density and total basal area (Figure 3.3). Mortality of unthinned ponderosa pine accounted for 7% of the decrease in total tree density and 12% of the decrease in total basal area while mortality of the other species accounted for 19% of the reduction in total tree density but only 3% of the decrease in total basal area in the treated area.

Actual ponderosa pine tree density in the treated area exceeded the projected outcome by 111-256%. Based on the pre-treatment sample data the projected outcome was to retain an average of 41.1 to 69.3 pines/ha comprised of 12.9 live presettlement pines/ha and 28.2 (150% of presettlement pine evidence) to 56.4 (300% of presettlement pine evidence) postsettlement replacement pines/ha. The post-treatment tree density averaged 146.3 pines/ha (Table 3.1) comprised of 10.4 live presettlement pines/ha and 135.7 postsettlement pines/ha. Pine density averaged 76.2 pines/ha when only thinned and burned plots were analyzed (Table 3.2) exceeding the projected residual density by only 10-85%.

One of the main goals of the restoration project was to avoid cutting and limit mortality of presettlement trees (trees with a center date older than 1870). However, a total of 3.0 presettlement trees/ha representing 0.18 m^2/ ha (PIPO=0.8 trees/ha and 0.1 m^2/ha, QUGA=2.2 trees/ha and 0.08 m^2/ha) were cut during the restoration treatment; none were cut in the control. A total of 13.5 presettlement trees/ha representing 1.1 m^2/ha (PIPO=1.7 trees/ha and 0.7 m^2/ha, QUGA=11.6 trees/ha and 0.4 m^2/ha, JUOS=0.2

trees/ha and 0.004 m^2) died in the treated area while a total of 2.8 presettlement trees/ha representing 0.5 m^2/ha (PIPO=1.3 trees/ha and 0.5 m^2/ha, QUGA=1.5 trees/ha and 0.06 m^2/ha) died in the control. However, 52.5 presettlement trees/ha representing 6.5 m^2/ha (PIPO=10.4 trees/ha and 4.8 m^2, QUGA= 41.1 trees/ha and 1.6 m^2/ha, JUOS= 1.0 trees/ha and 0.07 m^2/ha) remained alive in the treated area and 45.6 presettlement trees/ha representing 6.1 m^2/ha (PIPO=13.8 trees/ha and 4.8 m^2, QUGA= 31.2 trees/ha and 1.26 m^2/ha, JUOS= 0.55 trees/ha and 0.05 m^2/ha) remained alive in the control between 1996/97 and 2003.

The oldest living tree found in the study area was a ponderosa pine that was alive in 1455 (no pith on core) and the oldest oak had a center date of 1645; both were alive in 2003. In the treated area, 1.2% of the living pines and 0.5% of the living oaks were 200 years or older in 1996/97 compared to 2.8% (pine) and 0.4% (oak) in 2003. Nineteen percent of the pines and 40% of the oaks 200 years or older died after treatment. In the control, 1.4% of the living pines were 200 years or older in 1996/97 compared to 1.3% in 2003; oaks that were 200 years or older comprised 0.3% of the living oaks in both time periods. Six percent of the pines and none of the oaks 200 years or older died by 2003 in the control. In sum, these results show that mortality of presettlement pines was only slightly greater in the treated area compared to the control despite thinning and burning in the treated area. Oaks were much more susceptible to mortality in the treated area where thinning and burning occurred compared to the control, and compared to pines in general.

The diameter distribution of the treated forest shifted toward the historical distribution (Figure 3.4). In 1870, ponderosa pines <30 cm dbh comprised only 6% of the total trees/ha in the treated area and only 8% of the total trees/ha in the control. In the treated area, ponderosa pines <30 cm dbh comprised 40% of the overall total trees/ha prior to treatment and comprised 24% of the overall total in 2003. Seventy-seven percent of the pines thinned were taken from diameter classes <30 cm at breast height. In the control, 60% of the trees were <30 cm dbh in both 1996/97 and 2003. The shift in diameter distribution was more pronounced when plots that were thinned only, burned only, or not treated were excluded from analysis. On thinned and burned plots, pines <30 cm comprised only 16% of the overall total trees/ha (Figure 3.5).

Ponderosa pine comprised the majority of snags and logs >30 cm throughout the study area. Snag densities increased slightly in the control (total snags >30 cm dbh increased 19%, total snags >50 cm dbh increased 40%) but increased substantially in the treated area (total snags >30 cm dbh increased 95%, total snags >50 cm dbh increased 45%) between 1996/97 and 2003; however, no significant differences were found between the treated area and control for any time period or size category comparisons (Table 3.3). Log densities increased slightly in the control (logs >30 cm dbh increased 4%, logs >50 cm dbh increased 30%), but decreased in the treated area (logs >30 cm dbh decreased 20%, logs >50 cm dbh decreased 27%) between 1996/97 and 2003 (Table 3.3). In 2003, density composition for logs >30 cm differed between the treated area and control (pseudo-F =3.9, P=0.03); however, no statistically significant differences were found for any other time period or size category comparisons.

Regeneration was highly variable: minimum per-plot regeneration density was zero in both the treated and control areas and in both time periods and standard errors were high relative to means (Table 3.4). Gambel oak and New Mexico locust (both species can reproduce by sprouting) dominated regeneration density throughout the landscape before and after treatment; conifers had substantially lower densities. Combining all three height categories (Table 3.4), average total regeneration decreased by 10% in the control and 5% in the treated area between the two time periods. For stems <2 m in height in the treated area, locust increased by 49% after treatment; however, oak decreased by 14% and ponderosa pine decreased by 74%. In 2003, density composition for stems <30 cm in height differed between the treated area and control (pseudo-F=2.7, P=0.048); however, no significant differences were found for any other time period or height category comparisons.

Canopy cover (Table 3.5) was not significantly different (Z=0.3, P=0.7) between the treated area and control in 1996/97, but was significantly reduced (Z=6.0, P<0.0001) in the treated area in 2003. Canopy cover in the treated area was reduced from a pre-treatment average of 54% to a post-treatment average of 32%; there was little change in the control which remained at approximately 55%. The 2003 control had areas that were relatively open (minimum per-plot value = 9%) and the 2003 treated area had areas that were relatively dense (maximum per-plot value = 78%); these values represented outliers in each data set. Canopy cover for plots that were thinned and burned decreased from a pre-treatment average of 54% to a post-treatment average of 25%. Basal area explained 47% (adjusted r^2) of the variation in canopy cover ($y = 0.0099x + 0.2051$; n=233).

Discussion

Were the Restoration Treatments Implemented as Planned?

Were areas slated for restoration actually treated; and if so, according to the planned activity and schedule?

The treated area in 2003 closely resembled the original plan written in 1995 to thin and burn 433 ha within the approximately 700 ha treated area by 1998. About 94% of the area originally planned for restoration treatments was treated in some manner by 2003. Of these 409 treated ha, 74% (304 ha) were thinned and burned, 4% (15 ha) were burned only, and 22% (90 ha) were thinned only. Progress was slower than expected due to conflicting concerns of environmental groups, ranchers, and forest workers (Fulé 2003), and operational delays (e.g., lack of bids on thinning contracts and small burning windows). There were some notable changes to the original plan. Beginning in 1997, a nested stand-level study was introduced (Waltz et al. 2003) in which two small patches (24 ha total) were set aside as controls within the original treatment area and were not treated. Other areas were burned only (not thinned) due to the presence of archeological sites or were thinned only, since they had not yet been burned by our measurement in 2003.

Were the projected tree densities achieved after the implementation of restoration treatments?

Although post-treatment ponderosa pine tree density was 111-256% greater than the projected outcome, exceeding the residual tree density may be a better outcome than undershooting it, because treatment units can always be re-entered for further thinning whereas recruiting trees takes many years. In contrast, Fulé et al. (2002*a*) reported that restoration thinning in a dwarf-mistletoe infected stand near Grand Canyon actually resulted in post-treatment pine densities lower than presettlement densities. The average of 146 pines/ha includes all plots; when only thinned and burned plots were analyzed tree density exceeded the projected residual density by only 10-85%. Pine density averaged 76 pines/ha (Table 3.2) on thinned and burned plots which falls within the range (43-138 pines/ha) of post-treatment pine density found in other restoration studies (Lynch et al. 2000, Fulé et al. 2002*a*, Waltz et al. 2003).

Despite the contract amendments to terminate oak cutting, an average of 32 oaks/ha (0.5 m^2/ha) or 10% of living oaks were thinned, including 10 oaks/ha (0.25 m^2/ha) greater than 15 cm dbh. Cut oaks were detected on 34% of the plots within the treated area and were widely dispersed, indicating that oak cutting continued after the decision to stop cutting oaks, probably for several reasons: First, the decision to stop cutting oak was made after contracts for approximately 150 ha of the treated area had already been agreed upon (R. Davis, pers. comm.). Second, since Gambel oak is a valuable fuelwood species, some additional trees were cut illegally (A. Wilkerson, pers. comm.). Finally, since more than two-thirds of the oak trees cut were <15 cm dbh, they may have been cut by operators to ensure the safe removal of ponderosa pine trees. Fewer than 3 trees/ha (0.2 m^2/ha) of pinyon or locust were cut during restoration treatments and no juniper trees were cut.

With the exception of the oak cutting, the thinning prescription was designed to retain the living trees of non-pine species with the assumption that prescribed fire would thin these species. Twenty percent of the density and 80% of the basal area of Gambel oak died following restoration treatments, indicating that a greater proportion of larger oaks were killed compared to smaller ones. Two-thirds of the density and 40% of the basal area of New Mexico locust died, mainly because these deciduous species are susceptible to burning, but both species also sprout prolifically after fire (Mast 2003).

Were presettlement trees cut or killed during treatments?

Presettlement trees are, for all practical purposes, irreplaceable in a stand or landscape (DellaSalla et al. 2004) and were not intended to be cut in this project. These "legacy" trees provide genetic and structural diversity to the ecosystem and take centuries to replace (Moore et al. 1999, DellaSalla et al. 2004). The goal of the thinning prescription was to avoid any cutting of presettlement trees of any species and to protect them during prescribed burning. BLM fire prescriptions set a goal of not exceeding 10% mortality of presettlement pines. For oaks, the goal was to avoid purposely igniting clumps.

Despite this, six percent of the presettlement pines/ha and 4% of the presettlement oaks/ha alive prior to treatment were cut. None of the five pines used in the preceding calculation had viable increment cores and were therefore determined to be presettlement in the reconstruction model based on size rather than age data, meaning that these trees may have been younger, black-barked, but large trees. No ponderosa pine with a confirmed pre-1870 center date was cut. In contrast, most of the oaks used in the above calculation had viable cores with a pre-1870 center date. While the cutting of any of the old trees is a concern, it appears that oak cutting may be a more pressing issue than pine cutting.

Thirteen percent of the presettlement pines/ha died in the treated area by 2003, slightly exceeding the maximum allowable mortality outlined by managers. In the same period, 9% of the presettlement pines/ha died in the control, indicating that presettlement pines in the control are almost as vulnerable to mortality as those trees exposed to thinning and burning prescriptions. Although mortality may be partly attributed to natural factors such as drought and bark beetles, a treatment-related cause of presettlement pine mortality is likely the heat effects of prescribed fire. Forty percent of presettlement pine mortality in the treated area occurred on lava soils, consistent with Fulé et al.'s (2002*b*) finding that burning on lava soils at Mt. Trumbull caused high mortality of presettlement pines. Oak mortality was skewed much more sharply toward the treated landscape: 21% of presettlement oaks/ha died in the treated area by 2003 compared to only 5% in the control. Recommendations to avoid burning on lava soils (Fulé et al. 2002*b*) and to crush slash to reduce scorch (Jerman et al. 2004) are being incorporated for future treatment design.

Were the projected snag and log densities achieved?

Since large snags and downed logs provide important habitat, organic matter, and nutrients (Reynolds et al. 1992, Chambers 2002), the project goals were to retain at least 75% of the snags >50 cm dbh and at least 50% of the logs >50 cm dbh following burning (T. Duck, pers. comm.). Two-thirds of the snags >50 cm dbh remained standing in 2003, below the goal, but new recruitment resulted in a 45% net increase in large snags (Table 3.3). Eight percent of the snags >50 cm dbh were cut and 8% were consumed. Sixty-five percent of the logs >50 cm dbh were retained, thus, implementation goals for large logs were achieved. Most of the changes in snag density were due to prescribed burning but snags were occasionally felled for safety reasons and there was at least one documented incident of illegal cutting (T. Duck, pers. comm.). Changes in log density were fully attributed to prescribed fire. Snag and log consumption was highly variable and linked to adjacent fuel loads (pers. observation).

Were the Treatments Effective?

Did treatments effectively restore the historical forest structure while preserving the oldest trees?

Although restoration treatments were not implemented perfectly, the desired goal of restoring open forest structure conditions while preserving most of the old trees has been achieved; however, future management will be necessary to maintain the desired future dynamics of the ecosystem. Total tree density was reduced by nearly half and basal area was reduced by more than one-third after treatment. However, density levels remained more than three times greater and basal area more than double the 1870 values (Table 3.1). Eighty-two percent of the 2003 treated area was classified as having a low stand density index (Figure 3.2) compared to 44% before treatment. The diameter distribution of ponderosa pine, while still skewed compared to reconstructed distributions, was reduced in the proportion of smaller trees (Figure 3.4f). Furthermore, density and basal area were even lower when untreated and partially treated plots were not included. Diameter distribution of ponderosa pine on thinned and burned plots was unimodally distributed, as was the case in 1870 (3.5a). Finally, although some presettlement trees died after treatment, the majority survived and will likely be less susceptible to disease and insect attack (Wallin et al. 2004).

Were tree regeneration densities adequate and were canopy cover values reduced to presettlement levels?

Current tree regeneration densities are probably more than sufficient to sustain the current species composition and desired forest structure, although patterns are spatially variable. The prolific regeneration of oak and locust could lead to deciduous dominance, as Barton (2002) observed following wildfire in southern Arizona. However, subsequent burns are expected to continue to thin these species and unlike Barton's (2002) site, where mature seed-reproducing trees were absent, at Mt. Trumbull the density of seed producing trees found in the treated area (7.3 >65 cm trees/ha) exceeds restoration targets (1-2 >65 cm trees/ha) suggested by Bailey and Covington (2002). Although ponderosa pine seedling regeneration was substantially reduced in the treated area, the "snapshot" inventory of 3.3 <30 cm pine seedlings/ha Table 3.4) indicates that pine seedling density was at least similar in magnitude to the level needed for maintenance in presettlement times of 3.6 seedlings/ha/decade found by Mast et al. (1999). Waltz et al. (2003) also noted a substantial reduction of ponderosa pine regeneration following restoration treatments at Mt. Trumbull. Ongoing and frequent monitoring of regeneration is necessary at the Mt. Trumbull landscape.

Canopy cover values (Table 3.5) in the 2003 treated area were more consistent with presettlement values found near Flagstaff, Arizona of 17 percent, estimated by Covington and Sackett (1986) and 22 percent, estimated by White (1985). Canopy cover values from this study were lower than those found on Rainbow and Powell Plateaus, two sites at Grand Canyon National Park that have had relatively intact fire regimes and serve as reasonable reference sites for Mt. Trumbull (Fulé et al. 2002*c*, Fulé et al. 2003). Post-treatment canopy cover values were consistent with those found in other ponderosa pine restoration studies (Fulé et al. 2002*a*, Waltz et al. 2003, and Fulé et al. 2005). Reduced canopy cover, tree density, and ladder fuels all have implications for changing fire behavior as well (Fulé et al. 2001*b*); these changes have been assessed in detail elsewhere (see Chapter 4).

Management Implications

This study provided the first detailed information regarding the implementation and effectiveness of landscape-scale ecological restoration treatments in a southwestern ponderosa pine forest. Although the treatments were not implemented perfectly, the overall goal of rapidly re-establishing ecosystem characteristics similar to the reference characteristics was achieved. After treatment, the treated area at Mt. Trumbull was structurally more heterogeneous and more similar to pre-1870 conditions than the untreated control. From related finer-scale studies, there is reason to expect that these changes will result in improved ecosystem function (Covington et al. 1997, Kaye et al. 2005), increased vigor of old and young trees (Feeney et al. 1998, Stone et al. 1999, Skov et al. 2004), improved resistance to disturbance agents such as bark beetles (Wallin et al. 2004) and fire (Fulé et al. 2001*a*, Chapter 4), sufficient regeneration (Bailey and Covington 2002), and increased productivity of herbaceous understory vegetation (Covington et al. 1997, Laughlin et al., in press, Moore et al., in press). However, treatments have also resulted in the loss of some old trees from prescribed fire activities (Fulé et al. 2002*b*, Jerman et al. 2004) and the spread of the invasive exotic *Bromus tectorum* (C. McGlone, pers. comm.). Wildlife effects documented at Mt. Trumbull have been mixed to date, with beneficial and negative aspects depending on the animal species and scale of study (Germaine and Germaine 2002, Battin 2003, Germaine et al. 2004, Waltz and Covington 2004).

Evolution of treatments over time is common in broad-scale, extended management projects, but it is rare to have access to detailed data from permanent plots to assess changes. Lessons from this study include, first, that ongoing monitoring can be very helpful in identifying problems. We determined relatively early that cutting of oaks and heat effects of burning were issues of concern. Second, even after identifying issues there can be an administrative lag until changes take effect. Oaks in thinning contract areas, for instance, were thinned even after the decision was made to stop. Third, our data have identified new areas on which to focus attention in restoration treatments. Old ponderosa pines were largely uninjured, but old oak trees had a high rate of mortality. Future projects should maintain pine protection while addressing oak survival more explicitly. Fourth, monitoring offers quantitative data on which to rest decisions about future treatments. The Mt. Trumbull area contains additional dense ponderosa pine forests. Managers planning future restoration treatments can draw upon these lessons for developing new treatments and modifying existing prescriptions. Finally, a key point is that even after nearly ten years of treatment and monitoring, the work is not finished and the ecosystem is not "restored". Ongoing monitoring, maintenance of the surface fire regime, and continued management to address ecological impacts of fragmentation, exotic species, recreational use, etc., will remain important indefinitely.

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Table 3.1 Tree density (trees/ha) and basal area (m²/ha) (live trees ≥2.5 cm dbh) at the Mt. Trumbull landscape in 1870 (reconstructed), 1996/97 (pre-treatment), and 2003 (post-treatment).

Year	Treatment	Total	PIPO	QUGA	JUOS	PIED	RONE
Density (trees/ha)							
1870	Control	**85.3** a (7.4)	**43.1** (4.8)	**40.9** (7.4)	**1.3** (0.5)	0	0
		0-205.2	0-141.8	0-203.2	0-20.0		
1870	Treated	**108.2** a (12.1)	**31.7** (3.4)	**75.2** (12.8)	**1.3** (0.8)	0	0
		10.0-543.9	0-110.9	0-523.7	0-46.6		
1996/97	Control	**932.8** a (102.4)	**681.6** (106.2)	**184.2** (39.3)	**15.8** (4.6)	**4.6** (2.7)	**46.5** (36.5)
		107.7-3557.7	0-3427.0	0-1214.7	0-170.2	0-132.6	0-2003.6
1996/97	Treated	**784.6** a (65.6)	**428.6** (53.6)	**313.9** (42.2)	**11.3** (5.6)	**16.2** (4.0)	**14.6** (6.3)
		10.0-2061.6	10.0-1651.3	0-1366.8	0-279.1	0-151.6	0-355.6
2003	Control	**873.5** a (91.1)	**644.8** (95.2)	**171.6** (37.1)	**15.3** (4.5)	**4.6** (2.7)	**37.2** (31.4)
		107.7-3336.6	0-3216.0	0-1172.4	0-170.2	0-132.6	0-1723.1
2003	Treated	**399.2** b (51.1)	**146.3** (28.5)	**221.9** (35.3)	**10.6** (5.7)	**11.2** (4.1)	**9.1** (6.5)
		10.0-1581.3	10.0-1421.1	0-987.3	0-289.8	0-151.6	0-396.3
Basal area (m²/ha)							
1870	Control	**10.9** a (1.0)	**10.6** (1.0)	**0.3** (0.1)	**0.1** (0.03)	0	0
		0-32.8	0-32.8	0-2.4	0-1.4		
1870	Treated	**8.2** b (0.9)	**7.7** (0.8)	**0.5** (0.1)	**0.009** (0.006)	0	0
		0.004-25.6	0-25.6	0-4.2	0.03		
1996/97	Control	**31.6** a (1.9)	**27.1** (2.2)	**4.0** (0.7)	**0.3** (0.1)	**0.02** (0.02)	**0.2** (0.1)
		6.5-65.7	0-63.0	0-18.8	0-4.4	0-1.0	0-7.0
1996/97	Treated	**32.6** a (1.7)	**25.6** (1.7)	**6.4** (0.8)	**0.3** (0.1)	**0.2** (0.1)	**0.2** (0.1)
		7.7-63.2	0.2-61.1	0-22.8	0-5.5	0-3.6	0-3.0
2003	Control	**32.7** a (1.9)	**28.2** (2.2)	**4.0** (0.7)	**0.3** (0.1)	**0.03** (0.02)	**0.2** (0.2)
		9.7-68.4	0-65.5	0-19.1	0-4.7	0-1.1	0-8.7
2003	Treated	**18.9** b (1.4)	**13.6** (1.2)	**4.9** (0.7)	**0.3** (0.1)	**0.1** (0.1)	**0.05** (0.37)
		0.7-50.6	0.3-40.0	0-19.9	0-6.0	0-3.0	0-2.8

PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*; JUOS: *Juniperus osteosperma*; PIED: *Pinus edulis*; RONE: *Robinia Neomexicana*. Statistics presented are the **mean** (standard error), and minimum-maximum. Total tree density and total basal area were tested for differences between treatments in each year using Wilcoxon 2-sample tests. Within each year, different letters indicate significantly different means (P<0.05). Control n=55; Treated n=61.

Table 3.2 Tree density (trees/ha) and basal area (m²/ha) (live trees ≥2.5 cm dbh) at the treated area in 2003 (post-treatment).

Year	Treatment	Total	PIPO	QUGA	JUOS	PIED	RONE
Density (trees/ha)							
2003	Thin & Burn	**213.3** (38.6)	**76.2** (15.0)	**123.5** (31.3)	**0.6** (0.4)	**1.4** (1.4)	**11.6** (11.3)
		10.1-944.9	10.0-472.8	0-781.9	0-10.2	0-50.1	0-396.3
2003	Thin Only	**834.7** (141.4)	**290.5** (107.8)	**502.8** (99.8)	**7.5** (3.7)	**27.9** (10.6)	**5.9** (3.6)
		80.3-1581.3	20.0-1421.1	10.0-987.3	0-40.2	0-123.4	0-40.6
2003	Burn Only	**523.7** (155.0)	**196.4** (70.8)	**304.5** (185.0)	**10.0** (5.8)	**2.6** (2.6)	**10.2** (10.2)
		172.7-928.0	51.0-390.7	0-826.0	0-20.1	0-10.2	0-40.8
2003	Untreated	**477.6** (117.2)	**198.7** (85.2)	**196.0** (60.9)	**49.8** (33.1)	**29.1** (19.4)	**4.0** (3.1)
		10.0-1170.9	10.0-700.6	0-480.3	0-289.8	0-151.6	0-30.0
Basal area (m²/ha)							
2003	Thin & Burn Only	**14.6** (1.5)	**11.4** (1.3)	**3.1** (0.8)	**0.01** (0.01)	**0.01** (0.01)	**0.1** (0.1)
		0.7-39.0	0.3-30.4	0-18.8	0-0.3	0-0.4	0-2.8
2003	Thin Only	**24.2** (3.0)	**15.7** (3.1)	**8.1** (1.4)	**0.2** (0.1)	**0.1** (0.1)	**0.1** (0.04)
		3.9-40.2	0.5-37.1	0.3-16.6	0-1.1	0-0.8	0-0.4
2003	Burn Only	**27.2** (2.7)	**18.2** (4.2)	**8.2** (4.3)	**0.6** (0.4)	**0.1** (0.1)	**0.2** (0.2)
		24.4-35.4	5.8-24.5	0-17.7	0-1.7	0-0.2	0-0.7
2003	Untreated	**24.4** (4.6)	**16.8** (4.6)	**6.0** (2.2)	**1.1** (0.7)	**0.4** (0.3)	**0.09** (0.07)
		8.6-50.6	0.3-40.0	0-19.9	0-6.0	0-3.0	0-0.7

PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*; JUOS: *Juniperus osteosperma*; PIED: *Pinus edulis*; RONE: *Robinia Neomexicana*. Statistics presented are the **mean** (standard error), and minimum-maximum. Thin & Burn n=35; Thin Only n=12; Burn Only n=4; Untreated n=10.

Table 3.3 Large snag and log densities (trees/ha) for trees ≥30 cm dbh and trees ≥50 cm dbh at the Mt. Trumbull landscape in 1996/97 (pre-treatment) and 2003 (post- treatment).

Year	Treatment	Total	PIPO	QUGA	JUOS	RONE
Large snag density (trees/ha)						
1996/97	Control snags ≥30 cm	**4.2** (0.9) a 0-21.2	**2.7** (0.8) 0-20.4	**1.1** (0.6) 0-21.2	**0.4** (0.3) 0-10.0	0
1996/97	Treated snags ≥30 cm	**4.3** (0.9) a 0-30.1	**2.3** (0.6) 0-20.1	**2.0** (0.7) 0-30.1	0	0
2003	Control snags ≥30 cm	**5.0** (1.1) a 0-40.8	**3.7** (1.0) 0-40.8	**0.9** (0.5) 0-21.2	**0.4** (0.3) 0-10.0	0
2003	Treated snags ≥30 cm	**8.4** (1.6) a 0-70.1	**6.5** (1.5) 0-70.1	**2.0** (0.8) 0-30.1	0	0
1996/97	Control snags ≥50 cm	**2.0** (0.7) a 0-20.0	**2.0** (0.7) 0-20.0	0	0	0
1996/97	Treated snags ≥50 cm	**2.2** (0.6) a 0-20.0	**2.0** (0.6) 0-20.0	**0.2** (0.2) 0-10.1	0	0
2003	Control snags ≥50 cm	**2.8** (0.9) a 0-30.6	**2.8** (0.9) 0-30.6	0	0	0
2003	Treated snags ≥50 cm	**3.2** (0.7) a 0-20.5	**2.8** (0.7) 0-20.5	**0.3** (0.2) 0-10.1	0	0
Large log density (trees/ha)						
1996/97	Control logs ≥30 cm	**11.6** (2.1) a 0-80.4	**11.0** (2.0) 0-80.4	**0.4** (0.3) 0-10.3	**0.2** (0.2) 0-10.0	0
1996/97	Treated logs ≥30 cm	**10.4** (1.7) a 0-50.4	**9.9** (1.7) 0-50.4	**0.5** (0.3) 0-10.1	0	0
2003	Control logs ≥30 cm	**12.1** (2.1) a 0-80.4	**11.2** (2.0) 0-80.4	**0.7** (0.6) 0-30.8	**0.2** (0.2) 0-10.0	0
2003	Treated logs ≥30 cm	**8.3** (1.5) a 0-40.3	**7.4** (1.5) 0-40.3	**0.7** (0.3) 0-10.1	0	**0.2** (0.2) 0-10.0
1996/97	Control logs ≥50 cm	**6.6** (1.4) a 0-50.2	**6.6** (1.4) 0-50.2	0	0	0
1996/97	Treated logs ≥50 cm	**8.1** (1.4) a 0-40.3	**8.1** (1.4) 0-40.3	0	0	0
2003	Control logs ≥50 cm	**8.6** (1.5) a 0-50.2	**8.6** (1.5) 0-50.2	0	0	0
2003	Treated logs ≥50 cm	**5.9** (1.2) a 0-40.0	**5.9** (1.2) 0-40.0	0	0	0

Dead trees in condition class 7 (broken below breast height, 1.37 m) and 8 (dead and down) are listed together as "logs". PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*; JUOS: *Juniperus osteosperma*; RONE: *Robinia Neomexicana*. There were no *Pinus edulis* snags or logs recorded for either size class. Statistics presented are the **mean** (standard error), and minimum-maximum. Total snag and log densities for both size classes were tested for differences between treatments in each year using Wilcoxon 2-sample tests. Within each year, different letters indicate significantly different means ($P<0.05$). Control n=55; Treated n=61.

Table 3.4 Regeneration density (stems/ha) at the Mt. Trumbull landscape in 1996/97 (pre-treatment) and 2003 (post-treatment).

Year	Treatment	Total	PIPO	QUGA	JUOS	PIED	RONE
Regeneration 0-30 cm in height (stems/ha)							
1996/97	Control	**2226.8** a (782.4)	**7.3** (5.1)	**2179.1** (783.9)	**3.7** (3.7)	**3.8** (3.8)	**33.0** (33.0)
		0-38774.5	0-200.6	0-38774.5	0-201.0	0-206.6	0-1817.6
1996/97	Treated	**2291.7** a (585.6)	**36.2** (15.2)	**2067.4** (582.8)	**13.3** (9.3)	**16.5** (9.8)	**158.2** (77.6)
		0-24320.7	0-600.8	0-24320.7	0-411.4	0-402.9	0-3814.1
2003	Control	**2290.4** a (760.6)	**7.4** (5.2)	**2040.4** (749.4)	**14.6** (8.8)	0	**228.0** (152.9)
		0-38976.4	0-206.2	0-38976.4	0-402.0		0-7876.1
2003	Treated	**1986.0** a (493.8)	**3.3** (3.3)	**1721.5** (488.5)	**3.3** (3.3)	0	**257.9** (107.2)
		0-20099.8	0-200.6	0-20099.8	0-204.0		0-5417.3
Regeneration 30 cm to 2 m in height (stems/ha)							
1996/97	Control	**1525.2** a (413.2)	**62.8** (23.0)	**989.7** (310.7)	**7.3** (7.3)	0	**465.4** (254.0)
		0-14742.4	0-826.6	0-10703.4	0-402.0		0-12722.9
1996/97	Treated	**1728.4** a (331.7)	**65.8** (19.2)	**1081.5** (286.0)	**3.5** (3.5)	**3.3** (3.3)	**574.3** (206.6)
		0-9016.2	0-601.9	0-9016.2	0-214.7	0-201.4	0-8089.5
2003	Control	**1086.0** a (288.2)	**88.3** (27.7)	**794.2** (279.4)	**7.3** (5.1)	0	**239.6** (103.3)
		0-11691.1	0-826.6	0-11478.5	0-201.0		0-4241.0
2003	Treated	**1849.7** a (421.1)	**23.0** (16.9)	**979.4** (349.3)	**3.4** (3.4)	**7.1** (5.0)	**836.9** (276.6)
		0-18414.7	0-1001.2	0-18414.7	0-205.7	0-233.2	0-10413.0
Regeneration >2 m in height and <2.5 cm dbh (stems/ha)							
1996/97	Control	**11.0** a (8.1)	0	0	0	0	**11.0** (8.1)
		0-402.0					0-402.0
1996/97	Treated	**6.6** a (6.6)	0	0	0	0	**6.6** (6.6)
		0-400.5					0-400.5
2003	Control	**22.3** a (16.4)	0	**7.7** (7.7)	0	0	**14.6** (14.6)
		0-801.4		0-425.1			0-801.4
2003	Treated	0 a	0	0	0	0	0

PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*; JUOS: *Juniperus osteosperma*; PIED: *Pinus edulis*; RONE: *Robinia Neomexicana*. Statistics presented are the **mean** (standard error), and minimum-maximum. Total regeneration densities for each height class were tested for differences between treatments in each year using Wilcoxon 2-sample tests. Within each year, different letters indicate significantly different means (P<0.05). Control n=55; Treated n=61.

Table 3.5 Canopy cover (measured by vertical projection) at the Mt. Trumbull landscape in 1996/97 (pre-treatment) and 2003 (post-treatment).

Year	Treatment	N (no. of plots)	Mean (%)	SEM (%)	Minimum (%)	Maximum (%)
1996/97	Control	55	56.5 a	2.4	3.1	90.6
1996/97	Treated	61	54.6 a	2.5	12.5	87.5
2003	Control	55	54.0 a	2.3	9.4	81.2
2003	Treated	61	31.7 b	2.2	3.1	78.1
1996/97	T&B	35	54.7	3.4	15.6	87.5
2003	T&B	35	25.1	2.4	3.1	59.4

Canopy cover was estimated for one plot based on another plot with similar density and basal area for 1996/97 Treated.
T&B are a subset of plots within the treated area that were both thinned and burned.
Canopy cover was tested for differences between treatments in each year using Wilcoxon 2-sample tests. Within each year, different letters indicate significantly different means (P<0.05). Statistical testing was not performed for T&B plots.

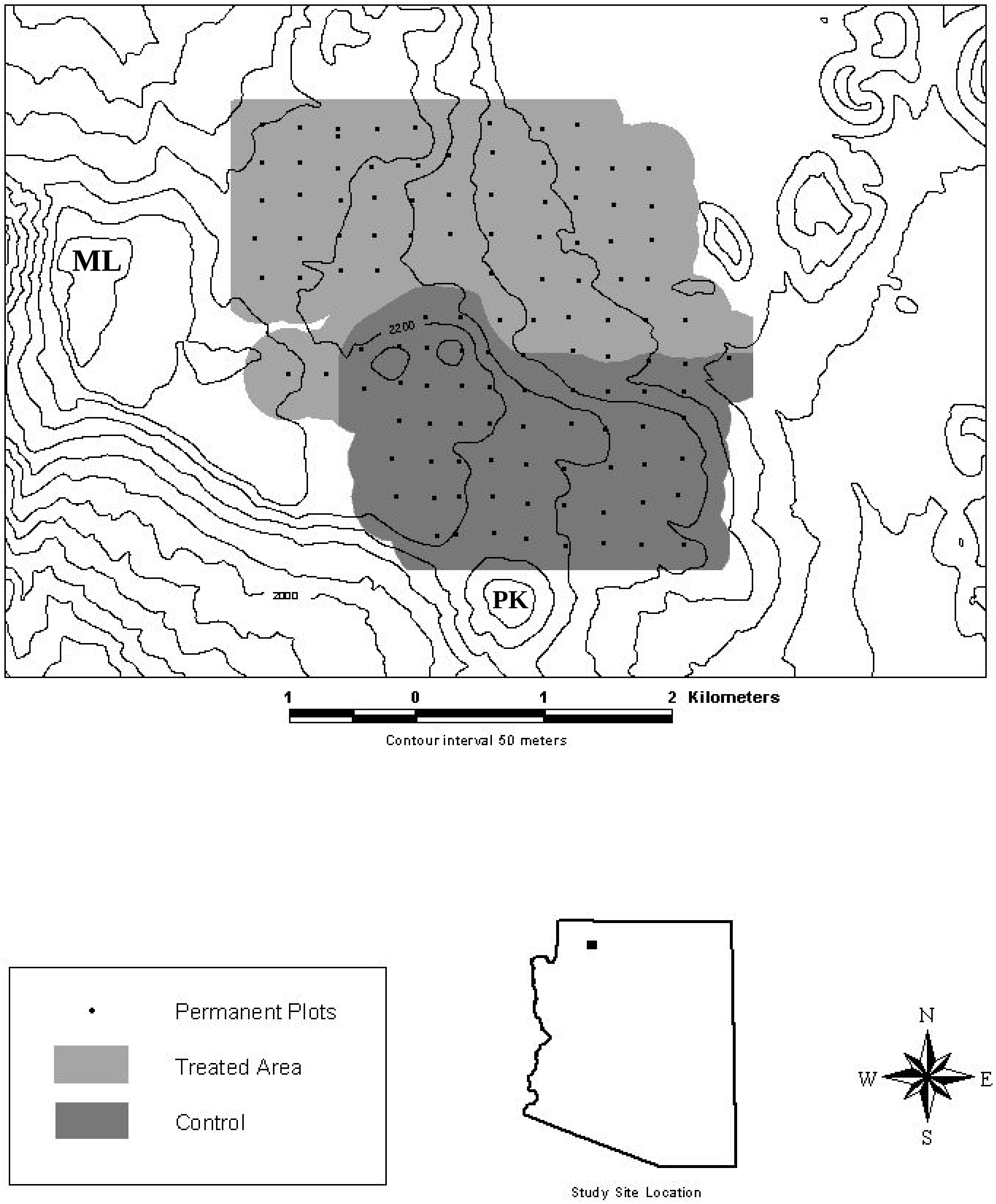


Figure 3.1 The map of the study site (~1200 ha) shows permanent plot locations. Mt. Logan (ML) is in the western part of the map; Petty Knoll (PK) is the mountain south of the control.

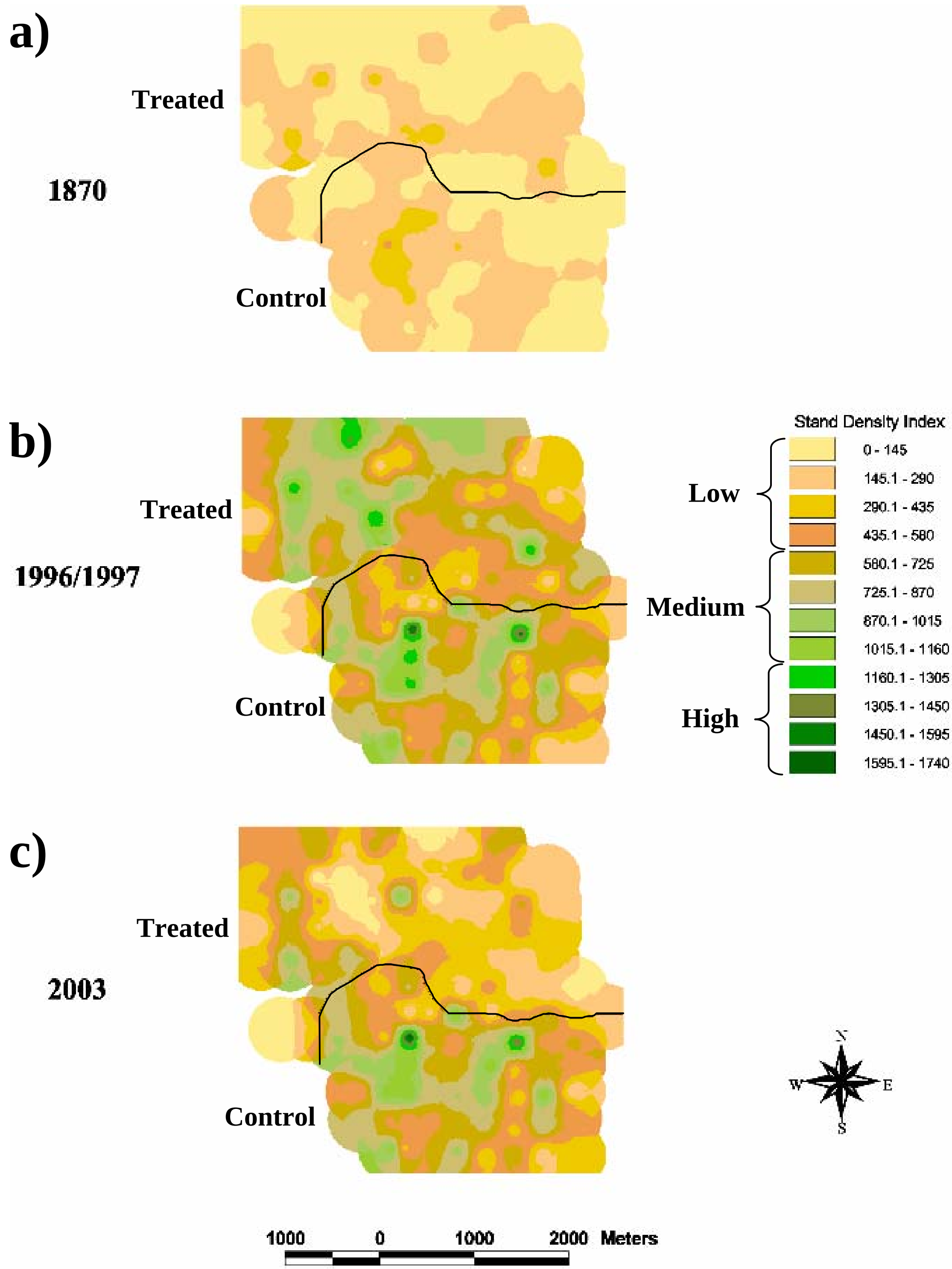


Figure 3.2 An interpolation of Stand Density Index (SDI) across the study area shows that 100% of the 1870 landscape had low density (SDI ≤580) compared to 44% prior to treatment and 62% after treatment. The control area is in the lower half of the landscape.

1870 | 1996/97 | 2003

Treated

a)

	TPH	BA (m²/ha)
PIPO L	32	8
Other L	77	1

b)

	TPH	BA (m²/ha)
PIPO L	429	26
Other L	356	7

c)

	TPH	%	BA (m²/ha)	%
PIPO L	143	18	13	40
Other L	245	31	5	15
PIPO D	29	4	2	5
Other D	74	9	1	4
PIPO C	257	33	11	34
Other C	38	5	0.6	2

Control

d)

	TPH	BA (m²/ha)
PIPO L	43	11
Other L	42	0.4

e)

	TPH	BA (m²/ha)
PIPO L	682	27
Other L	251	5

f)

	TPH	%	BA (m²/ha)	%
PIPO L	601	69	28	84
Other L	199	23	4	13
PIPO D	44	5	0.7	2
Other D	29	3	0.4	1

Figure 3.3 Stand visualization system (SVS) views of one hectare sized plots show that average pre-treatment density and basal area (b and e) were higher than in 1870 (a and d) and were reduced by 2003 in the treated area (c) while little change occurred in the control (f). Percentages in 2003 show the proportion of pre-treatment live total trees/ha and total basal area/ha that remained alive (L), died (D), or was cut (C) for ponderosa pine (PIPO) and the other four species combined (Other).

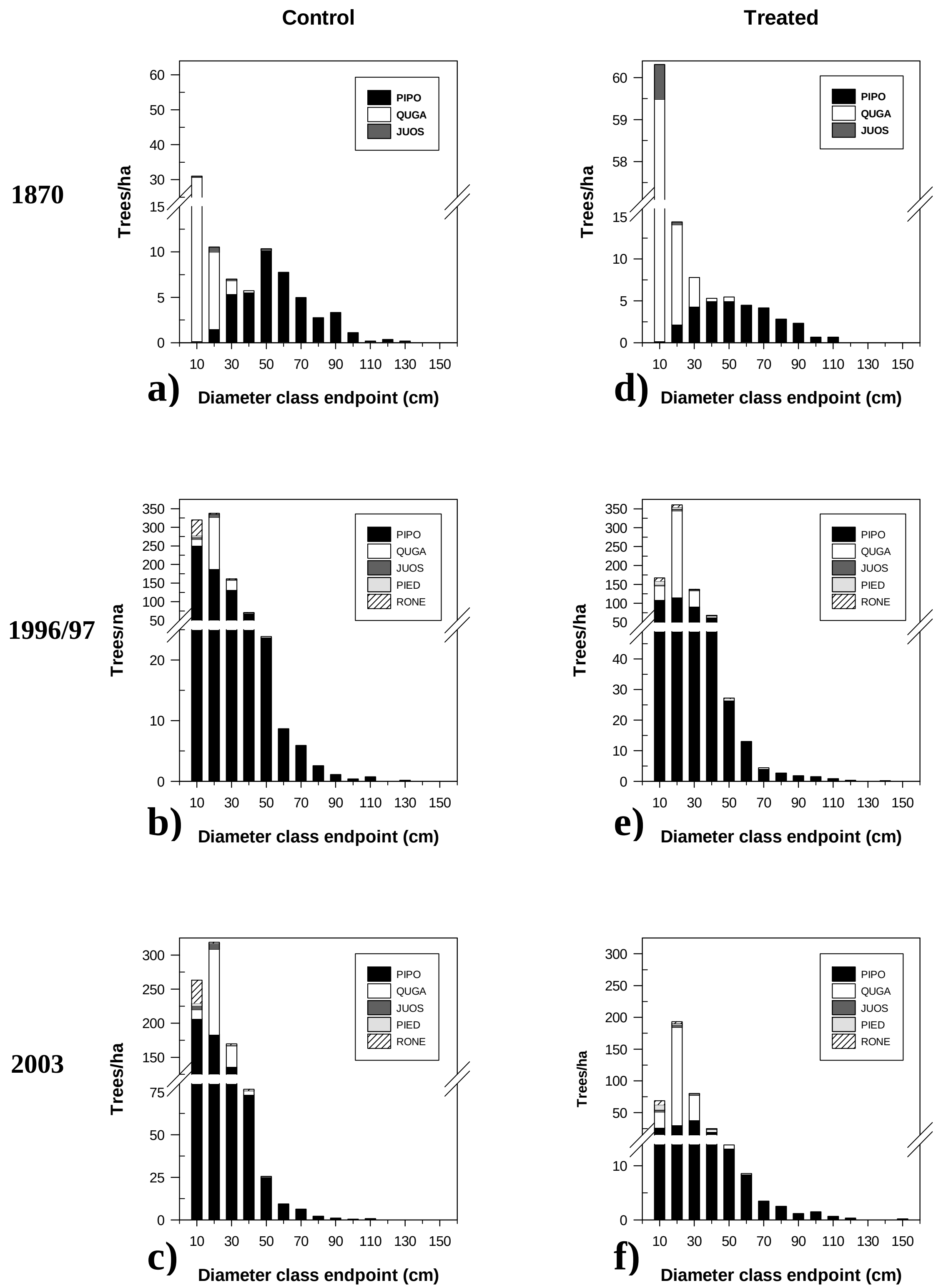
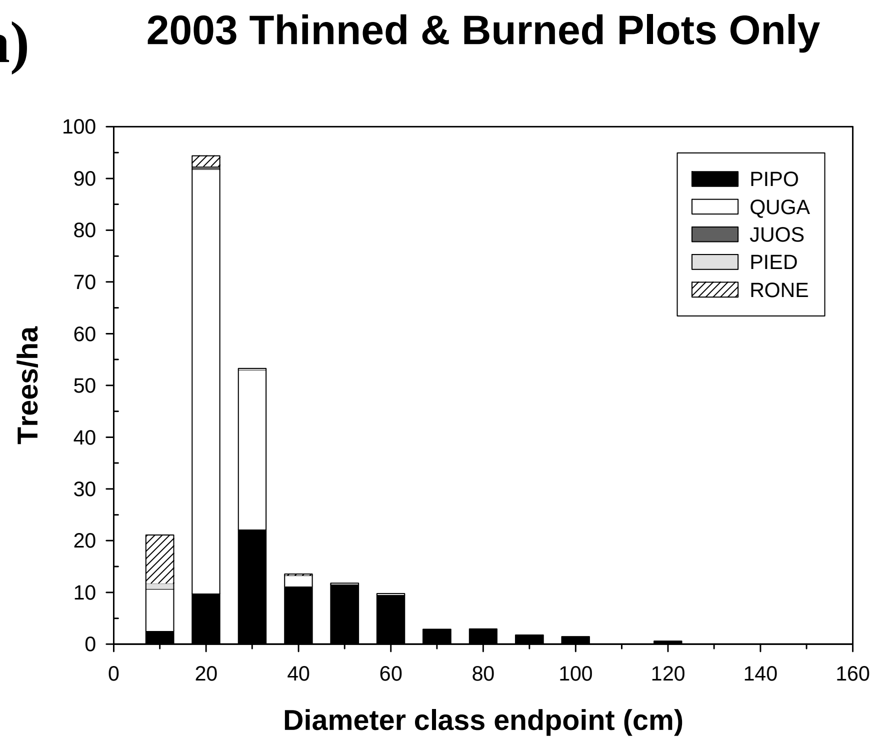


Figure 3.4 Diameter at breast height (dbh) distributions for the control and treated areas at Mt. Trumbull for 1870 (reconstructed at dbh) (a and d), 1996/97 (pre-treatment) (b and e), and 2003 (post-treatment) (c and f). Note scale differences between time periods.

a)



b)

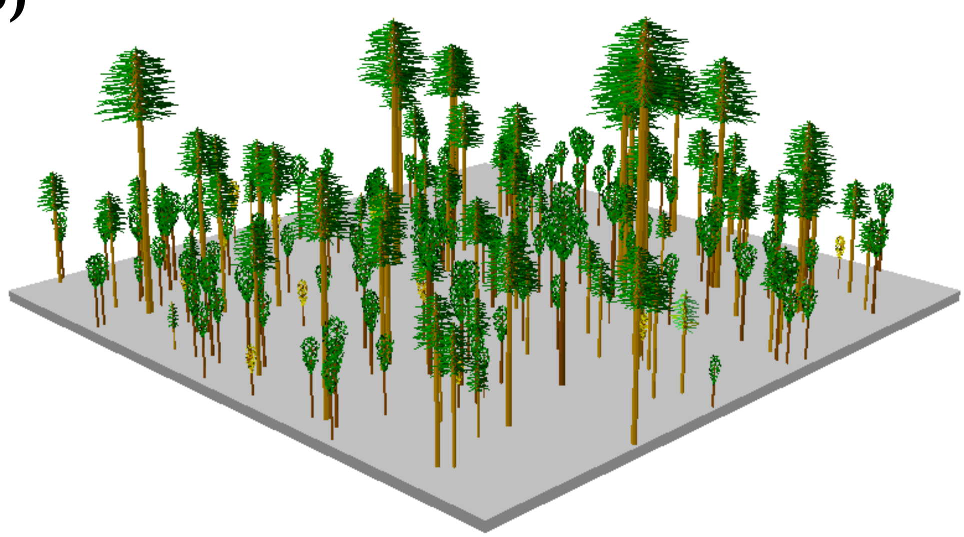


Figure 3.5 The diameter distribution (a) for the subset of thinned and burned plots within the treated area resembles the 1870 diameter distributions shown in Figure 3.4a and 3.4d; and, a stand visualization system (SVS) view (b) of a one hectare sized plot more closely resembles the average 1870 conditions shown in Figure 3.3a and 3.3d.

Chapter 4

Landscape-Scale Changes in Canopy Fuels and Potential Fire Behavior Following Ponderosa Pine Restoration Treatments

Abstract

We evaluated canopy fuels and potential fire behavior changes following landscape-scale forest restoration treatments in a ponderosa pine forest at Mt. Trumbull, Arizona. The goal of the project was to restore historical forest structure by thinning and burning, thereby reducing canopy fuels and minimizing the potential for active crown fire. We measured 117 permanent plots before (1996/97) and after (2003) treatments. The plots were evenly distributed across the landscape and represented an area of approximately 1200 ha, about half of which was an untreated control. We compared canopy fuel estimates using three different methods to assess whether fire behavior modeling outputs were sensitive to the choice of canopy fuel equation. Restoration treatments decreased canopy fuel load (CFL) by 43-50% from 7.7-18.3 Mg/ha to 4.4-9.1 Mg/ha (the range of values reflects the different canopy fuel equations) and decreased canopy bulk density (CBD) by 42-61% from 0.038-0.172 kg/m^3 to 0.022-0.67 kg/m^3 in the treated area, while slight increases occurred in the control. Canopy base height (CBH) averaged 0.4 m higher in the treated area than the control and 1.1 m higher when unburned plots within the treated area were excluded from analysis. We applied two simulation models to estimate potential fire behavior: FlamMap and Nexus. These models differ in several important features but predicted outcomes were consistent; under extreme drought and

wind conditions, the proportion of the landscape susceptible to active crown fire and the mean patch size of these areas were both reduced in the treated area. In contrast, the models show little change in active crown fire susceptibility in the control over the same time period. We conclude that the restoration treatments have successfully addressed the project goals of reducing canopy fuels and the potential for active crown fire.

Key Words: ecological restoration, fire behavior, FlamMap, canopy fuels, landscape-scale, Mt. Trumbull, Nexus, ponderosa pine

Introduction

The increase of stand-replacing crown fires in ecosystems that historically supported frequent surface fire regimes is a major ecological concern (Covington 2000, Allen et al. 2002). Many restoration and fuel treatment projects have been implemented throughout the western United States to restore natural ecosystem structure and function and to reduce the threat of stand-replacing crown fires (Scott 1998, Lynch et al. 2000, Fulé et al. 2001*a*, Stratton 2004). Such studies that examine treatment effects on fire behavior or severity can be classified into three categories: experimental, observational, and modeling. Experimental studies test fire behavior by purposely igniting fires and examining the effects during and after the burn. Although researchers have deliberately ignited crown fires to study their properties in certain isolated settings (Alexander et al. 2004), most experimental studies are focused on effects of relatively low-intensity fires (Weaver 1957, Covington et al. 1997, Fulé et al. 2002*a*) because intentionally lighting large, stand-replacing crown fires is difficult to justify. Observational studies examine the effects of wildfires after they occur. Pollet and Omi (2002), Martinson and Omi (2003), Graham (2003), and Cram and Baker (2003) showed that treated stands generally showed lower fire severity, although treatments did not necessarily preclude severe burning or prevent the passage of landscape-scale crown fires. Following the 2002 Rodeo-Chediski fire in Arizona, Finney et al. (2005) used satellite imagery and Strom (2005) used ground data to show that burning and/or cutting + burning treatments substantially reduced fire severity. Such observational approaches are essential for measuring real-world effects of treatments, but the scope of inference of the approach remains limited by lack of pre-fire data, randomization, and replication. The final technique, fire behavior modeling, is the most removed from actual fire behavior but the most flexible for testing alternative scenarios of stand development, treatments, or weather conditions. Various studies have used models to evaluate potential fire behavior after restoration or fuel treatments at scales ranging from stands (Stephens 1998, Fulé et al. 2001*a* & 2001*b*, Fulé et al. 2002*a*, Faiella 2005) to landscapes (Fiedler and Keegan 2003, Fulé et al. 2004, Stratton 2004).

Wildland fire is classified into ground (sub-surface), surface, or crown categories based on where in the fuel strata burning occurs (Pyne et al. 1996). Crown fires are further subdivided into three types: passive, active, and independent (Van Wagner 1977). Passive crown fire, or torching, occurs when fire transitions from the surface and ignites the lower canopy. The windspeed at which torching is initiated, the "torching index," is largely a function of canopy base height. Active crown fires burn the entire surface/canopy fuel complex, depending primarily on the bulk density of foliage and fine twigs in the canopy. Independent crown fires, or active crown fires that do not rely on surface fire, are extremely rare and not considered further here. Since passive and active crown fire behavior are linked to different canopy fuel variables, it is possible to encounter a situation where passive crown fire is not predicted to occur, due to a high canopy base height, but active crown fire could occur, due to high canopy bulk density. Scott and Reinhardt (2001) described this hysteresis as a "conditional" surface fire; active crown fire could occur on the condition that canopy burning entered the stand from outside, otherwise surface fire would occur.

Canopy fuels are a crucial input for models that predict crown fire but they are rarely measured directly. Brown (1978) provided allometric equations developed in the northern Rocky Mountains for ponderosa pine that have been widely applied. In Arizona, Fulé et al. (2001*a*, 2004) applied locally developed allometric equations that predicted less canopy fuel, and hence lower canopy bulk density, than would have been predicted by Brown's (1978) equations. Cruz et al. (2003) developed stand-level equations to predict canopy fuels based on tree density and basal area. Because these three approaches differ, the selection of a canopy fuel modeling approach may affect fire behavior model results.

Deterministic semi-empirical fire behavior models based on Rothermel's (1972) surface fire model, coupled with canopy initiation and spread models, are widely used in fire behavior analysis. FARSITE, a GIS-based system, uses terrain, fuels, and weather inputs to simulate the growth, spread, and behavior of wildland fires (Finney 1998). The variant FlamMap, adapted for assessing fuel hazards, uses most of the same inputs as FARSITE but predicts potential fire behavior simultaneously for each individual pixel on the raster landscape (Finney, in preparation). Nexus, another hazard model, uses plot- or stand-level data to predict potential fire behavior (Scott and Reinhardt 1999, 2001).

FlamMap and Nexus differ in crown fire outputs provided, with FlamMap simulating only passive and active crown fire, while Nexus also provides estimates of conditional surface fire.

Initiated in 1995, the Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project aimed to restore forest structure and ecosystem processes within the historical range of variability that occurred in the area prior to 1870 (Moore et al. 1999, Chapter 3). Additional goals of the project were to reduce fuel loads, disrupt fuel continuity, and reduce the risk of stand-replacing crown fires by implementing landscape-scale mechanical thinning followed by prescribed surface fire (Moore et al. 2003). In this study, we used fire behavior models to evaluate the effect of landscape-scale restoration treatments on crown fire hazard. Our goals were to: 1) compare three common canopy fuel estimation approaches using data collected in this study; 2) compare the output from FlamMap and Nexus; and 3) apply these analyses to assess the effectiveness of landscape-scale restoration treatments on reducing crown fire hazard.

Methods

Study Area

Mt. Trumbull is located in the Uinkaret Mountains on the Arizona Strip in the Grand Canyon-Parashant National Monument and managed by the Bureau of Land Management (BLM). Vegetation in the study area (elevation 2,000 to 2,250 m) is comprised of ponderosa pine (*Pinus ponderosa* P. & C. Lawson var. *scopulorum* Engelm.) and Gambel oak (*Quercus gambelii* Nutt.), with Utah juniper (*Juniperus* *osteosperma* [Torr.] Little), pinyon (*Pinus edulis* Engelm.), New Mexico locust (*Robinia neomexicana* Gray) and several shrubs occurring throughout the area. Soils are derived from basaltic parent material. The two main soil types found in the study area are the Wutoma-Lozinta complex which consists of ashy-skeletal over fragmental or cindery, mixed, mesic Vitrandic Haplustepts, and Sponiker soils, classified as fine, smectitic, mesic Pachic Argiustolls (Natural Resources Conservation Service 2004).

Native Americans, who named the mountains the Uinkarets or "region of pines," inhabited the Mt. Trumbull area for millennia before it was settled by Euro-Americans circa 1870 (Altschul and Fairley 1989). Records from fire-scarred trees suggest that relatively open forest structure conditions were maintained by a frequent fire regime prior to Euro-American settlement in 1870 (Waltz and Fulé 1998, Heinlein et al. 1999, Fulé, unpublished data).

Annual precipitation at Nixon Flats (elevation 1,981 m, approximately 3 km NE of study site) averaged 47.2 cm with an average January temperature of 1°C and an average July temperature of 21°C between January 1992 and December 2003 (Western Regional Climate Center 2005). Annual precipitation at Mt. Logan (elevation 2,195 m, approximately 2 km SW of study site) averaged 31.2 cm with an average January temperature of -1°C and an average July temperature of 20°C between January 1986 and December 2003 (Western Regional Climate Center 2005). Most precipitation occurs in winter and during summer monsoon storms; spring and fall are relatively dry.

About half of the approximately 1200 ha study landscape (Figure 4.1) is a contiguous, "untreated", densely-treed area (hereafter "control area" or "control"). The other half, hereafter "treated area", is adjacent to the control. Restoration treatments were carried out between 1996 and 2003 (Moore et al. 2003) and are described in detail in chapter 3. Some untreated areas remain within the treated area boundary, such as controls for other experiments, or operationally inaccessible areas, ranging from approximately 10 to 40 hectares.

Field Methods

Prior to treatment in 1996 and 1997, we installed 117 permanent plots on a 300 meter grid (Figure 4.1) throughout the Mt. Trumbull landscape as part of a before-after-control-impact (BACI) study design (Stewart-Oaten and Bence 2001); all plots (55 control, 61 treated, 1 partially treated, excluded from analysis) were remeasured in the summer of 2003. The plots were adapted from the National Park Service's Fire Monitoring plots (Reeberg 1995, NPS 2003), with modifications to collect dendroecological data for reconstruction of historical forest structure. Sampling plots were 0.1 ha (20 x 50 m) in size, oriented with the 50-m sides uphill-downhill to maximize sampling of variability along the elevational gradient and to permit correction of the plot area for slope.

Overstory trees, those larger than 15 cm diameter at breast height (dbh) were measured on the entire plot (1000 m^2) and trees between 2.5-15 cm dbh (pole-sized trees) were measured on one quarter-plot (250 m^2); all trees were tagged and species and dbh were recorded. Total height was measured for pole-sized trees but not for overstory trees during the pre-treatment measurement; total height and crown base height were measured for all trees in 2003. All overstory and pole-sized trees were also mapped within the 1000 m^2 plot. Ponderosa pine trees were considered potentially presettlement if dbh ≥37.5 cm or if bark was yellowed (White 1985). Trees of all other species were considered potentially presettlement if dbh ≥17 cm dbh (Barger and Ffolliott 1972). Tree cores were collected at 40 cm above ground level for all potentially presettlement trees and for a random 10% subsample of all other live trees ≥2.5 cm to determine past size, as described below. Canopy cover measured by vertical projection (Ganey and Block 1994) was recorded at 3 m intervals along the two 50-m sidelines of each plot for a total of 32 points per plot. Post-treatment measurements on plots coincided as closely as possible to the original day and month of the original measurement.

Reconstruction Methods

Tree increment cores were surfaced and crossdated (Stokes and Smiley 1968) using locally developed tree-ring chronologies. Rings were counted on cores that could not be crossdated, especially young trees and junipers. Additional years to the center were estimated using a pith locator (concentric circles matched to the curvature and density of the inner rings) for cores without a pith (Applequist 1958).

We reconstructed forest structure using dendroecological methods described in detail by Fulé et al. (1997) and Mast et al. (1999). We reconstructed diameter for all living trees by subtracting the radial growth since 1870 measured on increment cores and estimated death date of dead trees based on tree condition class using diameter dependent snag decomposition rates (Thomas et al. 1979, Rogers et al. 1984). We performed a sensitivity analysis by using the 25th, 50th, and 75th percentile decomposition rates to examine the effect of slower or faster decomposition on estimates of death date and 1870 structure. Less than ±1% change in reconstructed forest structure occurred during this analysis, so the 50th percentile reconstruction was used in this study.

Forest structure reconstruction methods were based on the assumption that evidence of all structures (i.e., snags, logs, stumps, stump holes) present in 1870 was intact, located, and correctly identified during the pre-treatment inventory. The probability that this occurred was relatively high given the absence of fire combined with the semi-arid environment limiting the decomposition of conifer wood (Fulé et al. 1997, Mast et al. 1999, Waltz et al. 2003), and because field crews were trained to identify the presence and species of presettlement structures. Moore et al. (2004) found that reconstruction field techniques in a similar environment and forest type were reliable within ±10% of tree density over ~90 years.

Fire Behavior Model Inputs

We used the following inputs for fire behavior modeling with both FlamMap (Finney, in preparation) and the Nexus Fire Behavior and Hazard Assessment System (Scott and Reinhardt 1999, 2001). Fuel model 9 (Anderson 1982), was used for all simulations. Fire weather extremes representing the 97th percentile of low fuel moisture for June from 34 years of data on the Kaibab National Forest (Tusayan weather station) were used in all simulations as described in Fulé et al. (2002*a*). These are very dry and windy conditions, representing the type of severe weather under which uncontrollable crown fires spread.

Canopy fuel load (CFL) and canopy bulk density (CBD) were estimated using three methods. The first method, described in detail in Fulé et al. (2001*a*), estimated CFL using locally developed allometric equations for foliage and fine twigs of ponderosa pine (Fulé et al. 2001*a*), Gambel oak (Clary and Teidemann 1986), and pinyon and juniper (Grier et al. 1992); oak equations were also used for locust. The second method used equations from Brown (1978) to calculate foliage and fine twigs for ponderosa pine, plus the non-pine-species equations used in the first method. In both of the first two methods, canopy volume (CV) was estimated using averages of maximum tree height (top of canopy) and crown base height (bottom of canopy). Canopy bulk density was calculated as CFL divided by CV for both methods. CV in 1870 was estimated using regression equations developed with data from Rainbow Plateau, Grand Canyon, Arizona, a nearby never-harvested reference site (Fulé et al. 2002*b*). The third method used equations from Cruz et al. (2003) to estimate CFL and CBD directly from tree density and basal area. We used the lowest quintile values (lowest 20%) of crown base heights on each plot as a model input to better represent actual conditions (Fulé et al. 2001*a*, 2002*a*). Foliar moisture content (FMC) was 80% unless noted otherwise.

FlamMap required additional GIS inputs, including elevation, slope, aspect, and canopy cover. The three topographic layers were derived from a digital elevational model (DEM). Canopy cover was calculated from plot data for 1996/97 and 2003 and was estimated using Rainbow Plateau regressions for 1870. Landscape patterns for each input layer were estimated using negative exponential interpolation between plots with 10 m resolution. We used windspeeds of 10 through 70 km/h, in 10 km/h increments. Wind azimuth was held constant at 225° to match the prevailing southwest wind direction at Mt. Trumbull during fire season. Each FlamMap run was saved as an ASCII file and then converted to raster files. The Patch Analyst (Version 3.0) extension for ArcView 3.3

(ESRI, Redlands, CA) was used to calculate percentage of the landscape and mean patch size based on the number of pixels in each fire type (i.e., surface, passive, or active). For Nexus, individual plot slope values were entered instead of DEM data. Nexus outputs were interpolated across the plot grid in ArcView to create maps. Percentage of the landscape was calculated based on percent of plots in each fire type.

Results

Canopy Fuels

Canopy fuel load (CFL) and canopy bulk density (CBD) values were relatively low over the entire study area in 1870, compared to later values (Table 4.1). Depending on which equations were used, CFL increased by 220-343% and CBD increased by 279-648% between 1870 and 1996/97 across the entire landscape. By 2003, treatment lowered CFL by 42% (Fulé et al 2001*a*), 48% (Brown 1978), and 61% (Cruz et al. 2003) and CBD by 43% (Fulé), 50% (Brown) and 50% (Cruz) in the treated area compared to slight increases in the control (Table 4.1). Canopy base height (CBH) was not measured prior to treatment, but after treatment CBH averaged 0.4 m higher in the treated area than the control and 1.1 m higher when unburned plots within the treated area were excluded from analysis (Table 4.2). Ponderosa pine CBH was higher in the treated area (4.8 m) than the control (3.7 m). Low quintile CBH was lowest in the control, higher in the overall treated area, and highest on thinned and burned plots (Table 4.2).

Potential Fire Behavior

FlamMap and Nexus modeling results for the three CBD levels showed that crown fire activity was correlated with CBD when modeled with constant windspeeds and FMC (Figure 4.2). The FlamMap simulations were very sensitive to CBD, showing no active crown fire at all when the lowest values (equations from Fulé et al. 2001*a*) were used and minimal active crown fire when intermediate values (equations from Brown 1978) were used. Therefore, we restricted the following analysis to results from the two fire behavior models using the highest CBD values, based on equations from Cruz et al. (2003).

FlamMap predicted that active crown fire would not occur within the study area in 1870 even with 70 km/h windspeeds (Figure 4.3). While only 5% of the 1870 landscape would support passive crown fire with 10 km/h winds, 64% of the landscape would support passive crown fire when windspeed was increased to 70 km/h (Figure 3.2). Mean patch size of areas that could support passive crown fire in 1870 increased from 2.1 ha with 10 km/h windspeeds to 52.9 ha with 70 km/h windspeeds.

In contrast to the 1870 condition, by 1996/97 areas that could support the initiation of active crown fire began to occur on the landscape with winds as low as 10 km/h (Figure 4.3). FlamMap predicted that 18% of the pre-treatment (1996/97) landscape would support passive crown fire even with 10 km/h windspeeds. When windspeeds were increased to 70 km/h, nearly 90% of the landscape was classified as either passive or active (see bar graph, Figure 4.3). In this scenario, FlamMap predicted that 13% of the landscape would burn with surface fire. However, because these areas had high CBH but also high CBD values, they likely would have been classified as conditional surface fire if FlamMap had this capability. Mean patch size of areas that could support passive crown fires increased from 3.0 ha with 10 km/h windspeeds to 4.6 ha with 70 km/h windspeeds in the pre-treatment landscape. Mean patch size of areas that could initiate active crown fires increased from 0.1 ha with 10 km/h windspeeds to 15.1 ha with 70 km/h windspeeds in 1996/97.

The 2003 FlamMap output (70 km/h windspeeds) indicated that the percent of the landscape susceptible to active crown fire initiation was reduced from 46% to less than 5% in the treated area; however, the model predicted that 69% of the treated area would still be able to support passive crown fire (Figure 4.3). Mean patch size of areas that could initiate active crown fires decreased from 14.9 hectares to 2.7 hectares in the treated area when modeled with 70 km/h windspeeds. In the control, the percent of the landscape susceptible to active crown fire initiation increased from 43% to 44% between 1996/97 and 2003 and mean patch size of areas that could initiate active crown fires increased from 14.4 ha to 29.5 ha when modeled using 70 km/h windspeeds. FlamMap predicted that 26% of the control would not support passive crown fire or initiate active crown fire when modeled using 70 km/h windspeeds; however, these areas would likely be classified as conditional surface fire if FlamMap could predict this situation (Figure 4.3).

Using Nexus to model potential fire behavior with the same model inputs used with FlamMap, we predicted that some active crown fire would occur within the study area in 1870 with windspeeds greater than 50 km/h, with up to 17% of the landscape supporting active crown fire when modeled with 70 km/h windspeeds (Figure 4.4). Like FlamMap, Nexus predicted that approximately two-thirds of the 1870 landscape would support passive crown fire with 70 km/h winds. Nexus predicted that 44% of the pre-treatment landscape would support passive crown fire and five percent would support active crown fire when modeled with 10 km/h windspeeds. When windspeeds were increased to 70 km/h, 80% of the pre-treatment landscape would support active crown fire. The 2003 Nexus output (70 km/h windspeeds) indicated that the percent of the landscape that could initiate or sustain active crown fire (i.e., conditional surface fire) was reduced from 82% to 48% in the treated area, however, the model predicted that less than 4% of the treated area would support surface fire. In the 2003 landscape, torching index (the windspeed necessary to initiate passive crown fire) and crowning index (the windspeed necessary to sustain active crown fire) were both greater in the treated area compared to the control (Figure 4.5). In the 2003 treated area, average torching index was three times greater and average crowning index was more than double pre-treatment levels (Table 4.3). Crown percent burned, rate of spread, heat per unit area, and average flame lengths were all reduced in the treated area when modeled using 2003 conditions.

Discussion

Canopy Fuels

Canopy fuel values are essential model inputs for both FlamMap and Nexus and it is important that canopy characteristics are estimated as accurately as possible (Scott and Reinhardt 2001). Values for CFL and CBD were highly variable depending on which equations were used (Table 4.1). Brown's (1978) equations always produced the highest value for average CFL, Fulé et al.'s (2001*a*) estimate was always lowest, and Cruz et al.'s

(2003) equation always produced values between Fulé et al. and Brown's CFL estimates. Cruz's CFL estimates were similar to Fulé's and exceeded them by only 0.1-12%; whereas, Brown's estimates exceeded Fulé's estimates by 79-163%. For CBD, Fulé's equation again produced the lowest estimate, but Cruz's estimate was highest and Brown's estimate was intermediate in all but one instance (1870, Treated) (Table 4.1). None of the CBD estimates were similar; Brown's estimates exceeded Fulé's estimates by 184-268% and Cruz's estimates exceeded Fulé's estimates by 153-491%. The percent change of CFL and CBD between time periods was also highly variable depending on which equation was used. For CFL, Fulé's estimates generally had the lowest and Brown's estimates generally had the highest percent change between time periods. Using the 1870 to 1996/97 CFL increase in the control as an example, Fulé's equation estimated a 187% increase compared to an increase of 315% by Brown and an increase of 208% by Cruz. For CBD, Fulé's estimates had the lowest and Cruz's estimates had the highest percent change between time periods.

Which equations should be used in southwestern ponderosa pine forests? Direct measurement of canopy fuels has recently been completed on a dense ponderosa pine plot (10-m radius) near Flagstaff, Arizona, approximately 160 km southeast of our study area (Scott and Reinhardt 2005). Prior to treatment, the plot's CFL was 9.3 Mg/ha and CBD was 0.17 kg/m^3; removal of 75% of the original basal area by thinning from below reduced these values to 2.7 Mg/ha and 0.057 kg/m^3 (Scott and Reinhardt 2005). These values for CFL are similar to those generated by the Fulé and the Cruz equations in the untreated 1996/97 landscape (7.7 to 9.8 Mg/ha, respectively; values with the Brown equations were much higher: 18.3 to 23.1 Mg/ha [Table 4.1]). In contrast, however, Scott and Reinhardt's (2005) CBD measurements were most similar to our CBD estimates using the Cruz's equation (0.172-0.226 kg/m^3). Post-treatment values were comparable to Scott and Reinhardt's (2005) thinned values following the same pattern as in the untreated forest.

Given the variety of ways in which these equations were developed, it would be difficult to speculate about the reasons for the differences. Assuming that Scott and Reinhardt's (2005) measured data are the most accurate available in northern Arizona, we conclude that the locally developed allometric equations presented by Fulé et al. (2001*a*) are appropriate estimates of the actual available canopy fuel biomass. However, the approach used by Fulé et al. (2001*a*) to calculate CFL by CV assumes that canopy fuels are evenly distributed throughout the canopy volume. Scott and Reinhardt (2005) illustrate that CBD varies through the canopy and they report the peak values, arguing that maximum CBD is the most important factor in assessing crown fire spread. If this logic is applied, then it would be appropriate to use the CBD equation presented by Cruz et al. (2003) in southwestern ponderosa pine forests because Cruz's equation produces CBD values closest to those reported by Scott and Reinhardt (2005). As canopy fuel measurement grows more sophisticated, estimates will become increasingly accurate. For now, however, it may be advisable for analysts to use several approaches, as we did in this study, in order to understand the sensitivity of outputs to changes in inputs.

Potential Fire Behavior

Fire behavior model outputs should be interpreted with caution. The purpose of modeling fire behavior was not to accurately estimate the behavior of an actual fire, but rather to use the output as a means of comparing potential fire behavior between the three time periods as well as between the control and treated areas in 2003 over a range of windspeeds. There are two major differences between the two models. First, in FlamMap, the model inputs are interpolated across the plot grid and results are calculated for each 10 x 10 m cell, whereas, in Nexus, inputs are calculated for each plot and outputs are interpolated across the landscape. Second, Nexus accounts for the situation known as the conditional surface fire (Scott and Reinhardt 2001); FlamMap does not.

We initially modeled fire behavior using FlamMap because it incorporated detailed topographical information (elevation, aspect, and slope) for each 10 by 10 m cell. Since our intent was to evaluate restoration treatment effectiveness on crown fire hazard, we used Cruz et al.'s (2003) CBD because Fulé et al.'s (2001*a*) and Brown's (1978) CBD produced little crown fire even under the most extreme conditions in FlamMap (Figure 4.2). Nexus was less sensitive to CBD and produced active crown fire even with the lowest CBD values estimated using the equations from Fulé et al. (2001*a*). Cruz et al.'s (2003) equation produced the highest percentage of active crown fire in both models.

In general, FlamMap and Nexus produced similar results: relatively low crown fire hazard in 1870, a marked increase in crown fire hazard by 1996/97, and decreasing crown fire hazard in the treated area by 2003 with little change in the control (Figure 4.3 and Figure 4.4). With FlamMap, increases in windspeed produced gradual increases in active crown fire, whereas with Nexus active crown fire increased more dramatically at a threshold windspeed of approximately 30-40 kph (see bar graphs in Figure 4.3 and Figure 4.4). Overall, Nexus always predicted more passive and active crown fire compared to FlamMap when modeled under the same conditions. For example, comparing the 1870 output for the two models, with 30 kph windspeeds, Nexus predicted that the percent of the landscape classified as passive crown fire was more than two times greater than FlamMap's prediction; similarly, with 70 kph windspeeds, FlamMap did not predict any active crown fire while Nexus predicted about 20%.

The most significant difference between the two models is that Nexus accounts for conditional surface fire, whereas FlamMap does not. In FlamMap, a pixel cannot be classified as active crown fire unless it is first classified as passive crown fire (Finney, in preparation), thus, the maps produced by FlamMap only show areas that can initiate active crown fire. Therefore, if torching index exceeded the crowning index for a given cell and could sustain active crown fire, FlamMap would consider it surface fire even though it should be considered conditional surface fire. FlamMap's inability to predict conditional surface fire was demonstrated by the large area in the 2003 control classified as surface fire even when modeled with extreme weather conditions (Figure 4.3). Based on plot data and photos, it was evident that this area had abundant canopy fuels that could sustain an already burning active crown fire, but because of insufficient ladder fuels (i.e., high CBH), would be unable to initiate active burning even with extreme weather conditions. We suspected that this area would have been classified as conditional surface fire if FlamMap had the ability to do so. The Nexus output revealed that torching index exceeded the crowning index for the plots in this area, thus, Nexus classified this area as active when modeled with 70 km/h windspeeds (Figure 4.4).

The final objective in this study, and perhaps the most important, was to use FlamMap and Nexus to evaluate the effectiveness of restoration treatments on crown fire hazard. Both models predicted a range of variability in fire behavior throughout the landscape, but both consistently predicted reduced crown fire hazard in the treated area compared to pre-treatment levels and compared to the control (Figure 4.3 and Figure 4.4). Although both models predicted large areas of passive crown fire in the 2003 treated area with high windspeeds, areas classified as active crown fire were limited to two small patches. Furthermore, the patterns observed in the 2003 treated area resembled the fire behavior predicted for the 1870 landscape for both models. Increased torching and crowning indices due to treatment (Table 4.3) were consistent with results from previous research on similar restoration treatments in northern Arizona (Fulé et al. 2001*a*, Fulé et al. 2001*b*, Fulé et al. 2002*a*, Faiella 2005). The model results were also consistent with on-the-ground fire behavior observations. In April 2000, the "EB3 Escape Fire" burned as active crown fire in an untreated control unit (Waltz et al. 2003) directly adjacent to our study site.

Management Implications

Many researchers have used models to estimate canopy fuels and predict potential fire behavior, but few studies compare the results of more than one model. All models are simplifications of reality and are based on certain assumptions. We suggest that fire behavior analysts use multiple modeling approaches when possible to better support their findings. Inclusion of conditional surface fire classification would be a useful addition to the FlamMap model.

This study provided the first evaluation of the effectiveness of landscape-scale restoration treatments on canopy fuels and crown fire hazard for the Mt. Trumbull landscape. Although canopy fuel estimates and fire behavior predictions varied depending on which models were used, all modeling scenarios resulted in substantially lowered canopy fuels and crown fire hazard in the treated area.

The Mt. Trumbull ecosystem will never be “fireproofed”. Some level of crown fire will likely occur in the future, particularly in untreated areas. Even within treated areas, passive crown fire may occur, especially during dry years. However, the overall management objective of reducing canopy fuels and crown fire hazard was achieved in treated areas. Maintenance of the surface fire regime will be vital to retaining open forest conditions and relatively low crown fire hazard into the future. The Mt. Trumbull area contains additional dense ponderosa pine forests. If these areas remain untreated, stand-replacing crown fires could cause large patches with high tree mortality which could potentially limit conifer regeneration (Barclay et al. 2004). Severe fires may even result in ecosystem conversion to shrubfields or grasslands (Savage and Mast 2005, Strom 2005). The Mt. Trumbull ponderosa pine ecosystem is not yet “restored”. However, restoration treatments have been successful at substantially reducing crown fire hazard and creating more sustainable forest conditions. Managers should continue to model crown fire hazard and monitor on-the-ground fire behavior as additional areas within the project are treated.

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Table 4.1 Canopy fuel load (CFL) and canopy bulk density (CBD) values at the Mt. Trumbull landscape in 1870 (reconstructed), 1996/97 (pre-treatment), and 2003 (post-treatment) using three different equations (Fulé et al. 2001*a*, Brown 1978, Cruz et al. 2003).

Year	Treatment	Fulé et al. (2001*a*)	Brown (1978)	Cruz et al. (2003)	Fulé et al. (2001*a*)	Brown (1978)	Cruz et al. (2003)
			CFL (Mg/ha)			*CBD (kg/m³)*	
1870	Control	**2.967** (0.279)	**5.412** (0.543)	**3.078** (0.300)	**0.013** (0.001)	**0.024** (0.002)	**0.031** (0.003)
		0-8.938	0-15.787	0-9.228	0-0.038	0-0.077	0-0.085
1870	Treated	**2.170** (0.232)	**3.888** (0.422)	**2.295** (0.247)	**0.015** (0.003)	**0.028** (0.006)	**0.023** (0.002)
		0.001-7.136	0.001-14.219	0-7.558	0-0.134	0-0.278	0-0.078
1996/97	Control	**8.528** (0.649)	**22.468** (2.018)	**9.492** (0.805)	**0.046** (0.004)	**0.122** (0.011)	**0.226** (0.027)
		0.642-20.119	0.642-63.344	0-24.062	0.007-0.116	0.007-0.365	0-0.872
1996/97	Treated	**7.708** (0.478)	**18.309** (1.526)	**8.657** (0.614)	**0.038** (0.003)	**0.093** (0.009)	**0.172** (0.134)
		1.258-19.005	1.429-55.742	0.096-21.714	0.009-0.098	0.006-0.289	0.003-0.564
2003	Control	**8.757** (0.629)	**23.052** (1.988)	**9.768** (0.784)	**0.047** (0.003)	**0.126** (0.011)	**0.224** (0.026)
		0.701-20.403	0.701-63.917	0-24.831	0.007-0.118	0.007-0.369	0-0.857
2003	Treated	**4.356** (0.365)	**9.105** (0.984)	**4.360** (0.418)	**0.022** (0.002)	**0.048** (0.006)	**0.067** (0.010)
		0.177-12.556	0.299-35.141	0.118-13.898	0.002-0.075	0.003-0.205	0.003-0.417

Statistics presented are the **mean** (standard error), and minimum-maximum. Control n=55; Treated n=61.

Table 4.2 Average and low quintile (LQ) canopy base height (CBH) (m) at the Mt. Trumbull landscape in 2003 (post-treatment).

Year	Treatment	Total		PIPO		QUGA		JUOS		PIED		RONE	
CBH (m)		Avg.	LQ	Avg.	LQ	Avg.	LQ	Avg.	LQ	Avg.	LQ	Avg.	LQ
2003	Control	3.5 (0.2)	1.8 (0.2)	3.7 (0.3)	1.7 (0.2)	2.1 (0.1)	1.7 (0.1)	1.3 (0.3)	1.2 (0.3)	0.8 (0.6)	0.8 (0.6)	2.4 (0.4)	2.1 (0.4)
2003	Treated	3.9 (0.3)	2.6 (0.3)	4.8 (0.3)	3.3 (0.3)	2.2 (0.1)	1.7 (0.1)	1.3 (0.2)	1.0 (0.2)	0.7 (0.2)	0.6 (0.2)	3.0 (0.6)	2.7 (0.7)
2003	T&B	4.6 (0.5)	3.2 (0.5)	5.7 (0.4)	4.4 (0.5)	2.1 (0.2)	1.7 (0.2)	1.6 (0.6)	1.6 (0.6)	0.6	0.5	1.7 (0.3)	1.3 (0.7)

Low quintile (LQ) values are the lowest 20% of crown base heights on each plot.
T&B are a subset of plots within the treated area that were both thinned and burned.
PIPO: *Pinus ponderosa*; QUGA: *Quercus gambelii*; JUOS: *Juniperus osteosperma*; PIED: *Pinus edulis*; RONE: *Robinia Neomexicana*. Statistics presented are the **mean** (standard error). Crown base height data was collected in 2003 only. Control n=51; Treated n=61; T&B n= 35.

Table 4.3 Fire behavior outputs predicted by Nexus at the Mt. Trumbull landscape in 1870 (reconstructed), 1996/97 (pre-treatment), and 2003 (post-treatment).

	1870		1996/97		2003		
Fire Behavior Output	**Control**	**Treated**	**Control**	**Treated**	**Control**	**Treated**	**T&B**
Crown percent burned	52.4	39.8	90.0	91.3	89.5	66.0	57.7
Rate of spread (m/min)	46.8	41.1	63.8	64.1	63.5	52.8	49.0
Heat/area (kJ/m²)	10038.1	8566.2	22395.2	21171.4	22867.4	12550.6	10621.8
Flame length (m)	9.2	7.1	21.5	20.9	21.8	12.4	10.3
Torching index (km/h)	27.6	22.0	16.8	9.4	16.8	27.1	33.6
Crowning index (km/h)	112.9	138.2	42.6	51.6	39.4	105.7	115.6

T&B are a subset of plots within the treated area that were both thinned and burned.
Control n=55; Treated n=61; T&B n=35.

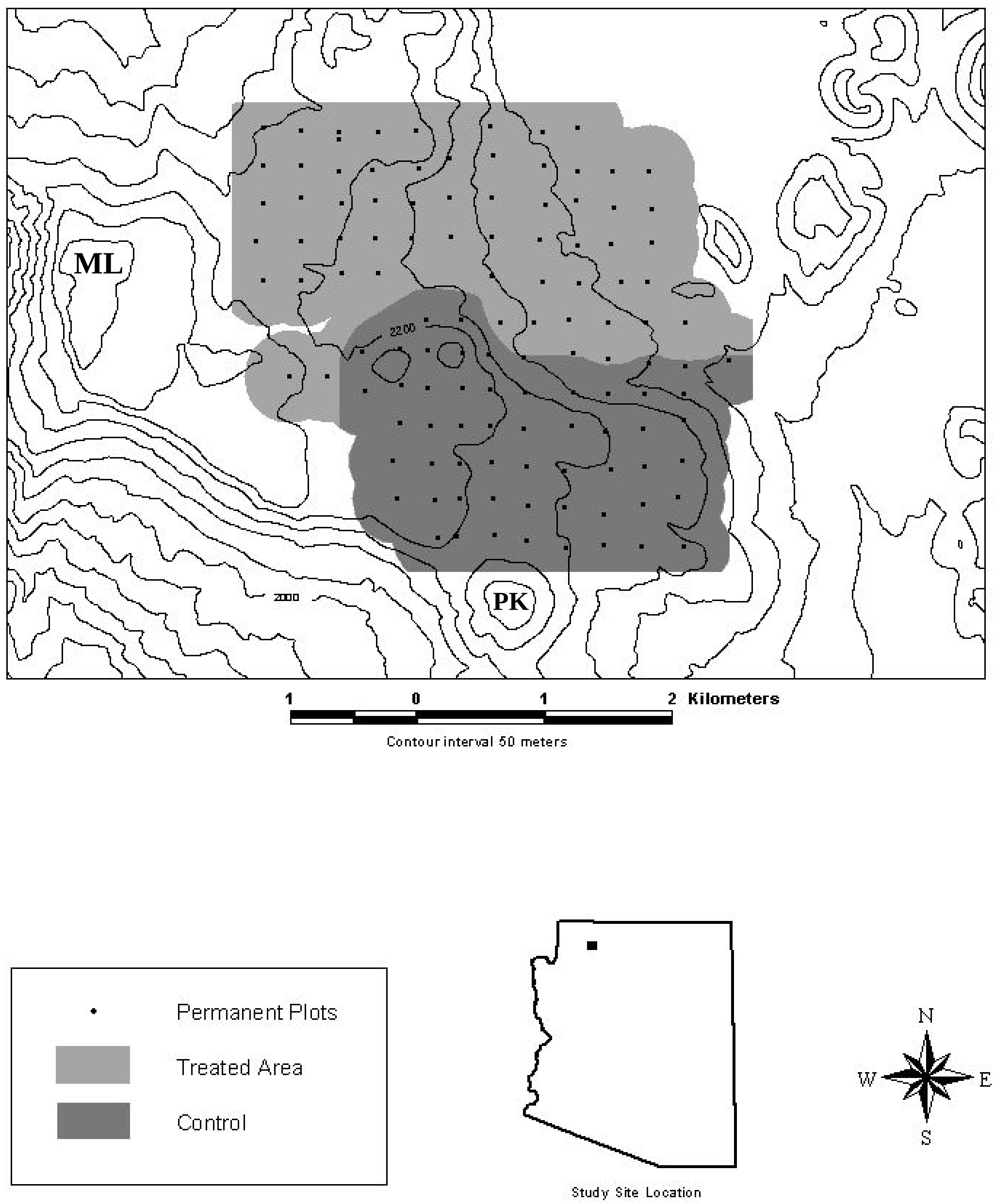

Figure 4.1 The map of the study site (~1200 ha) shows permanent plot locations. Mt. Logan (ML) is in the western part of the map; Petty Knoll (PK) is the mountain south of the control.

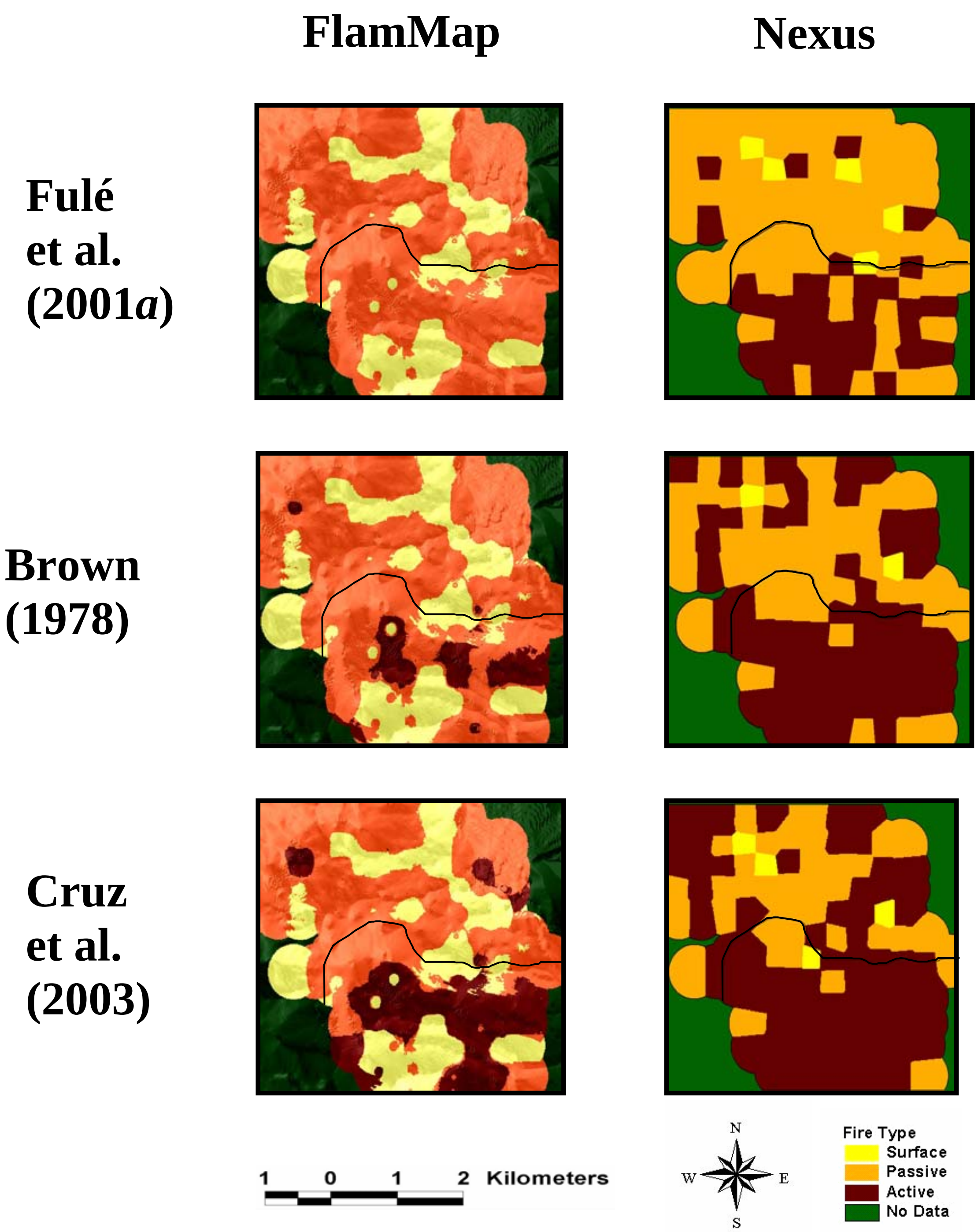

Figure 4.2 FlamMap and Nexus output modeling results show that FlamMap is not responsive to the lower CBD values produced using Fulé et al.'s (2001*a*) equations. Cruz et al.'s (2003) equations produce the highest percentage of active crown fire throughout the landscape in both models. Model inputs were 70 km/h windspeeds, 80% foliar moisture content, low quintile CBH, 2003 conditions. The control is in the lower half of the landscape.

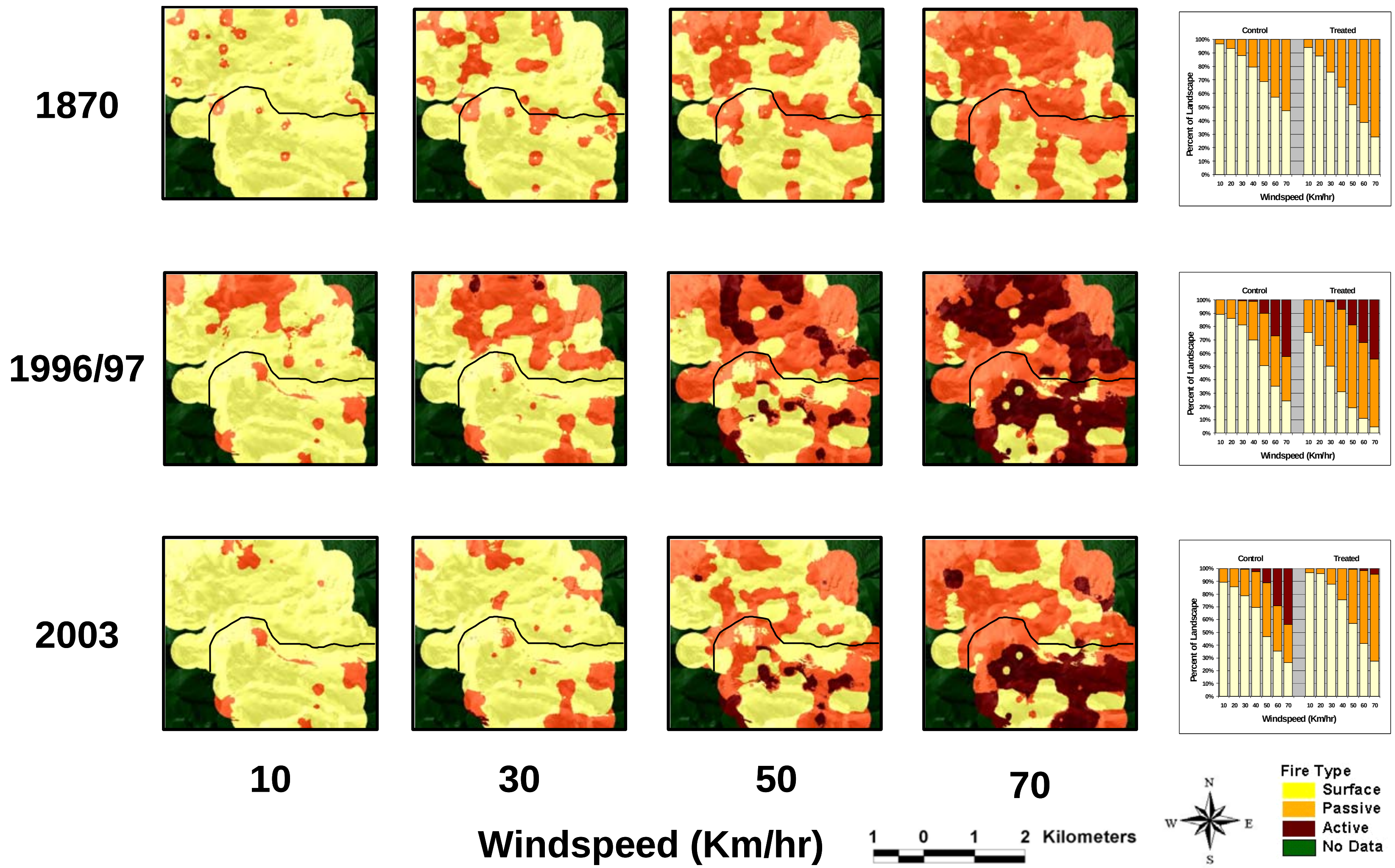

Figure 4.3 An interpolation of potential fire behavior across the study area using FlamMap shows that under extreme conditions (80% foliar moisture content and 70 Km/hr winds) active crown fire initiation would occur on 0% of the 1870 landscape compared to 44% prior to treatment (1996/97) and 21% after treatment (2003). The control is in the lower half of the landscape. Model inputs used were low quintile CBH, CBD using Cruz et al. (2003), and 80% foliar moisture content.

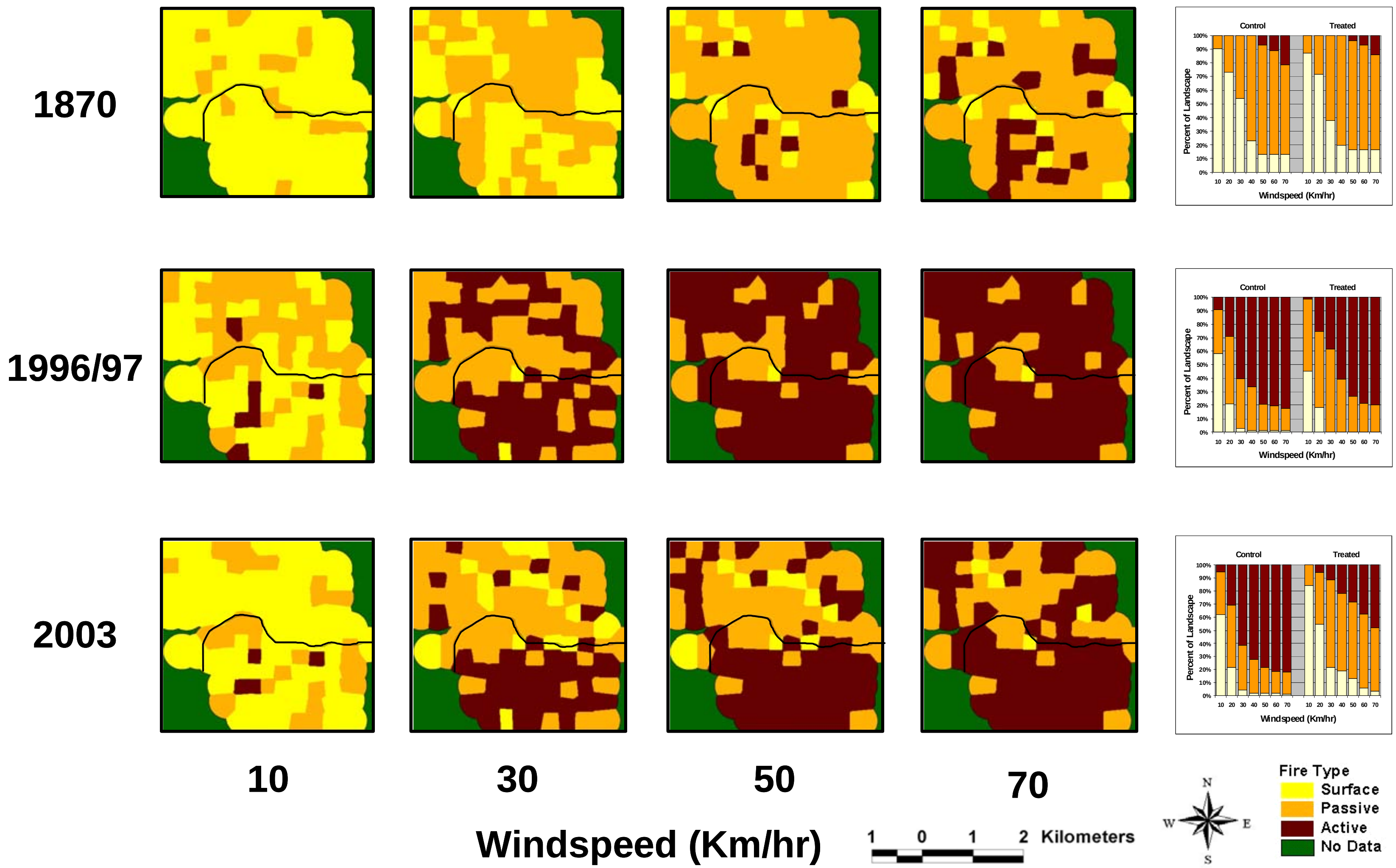

Figure 4.4 An interpolation of potential fire behavior across the study area using Nexus shows that under extreme conditions (80% foliar moisture content and 70 Km/hr winds) only 17% of the 1870 landscape would support active crown fire compared to 81% prior to treatment (1996/97) and 63% after treatment (2003). The control is in the lower half of the landscape. Model inputs used were low quintile CBH, CBD using Cruz et al. (2003), and 80% foliar moisture content.

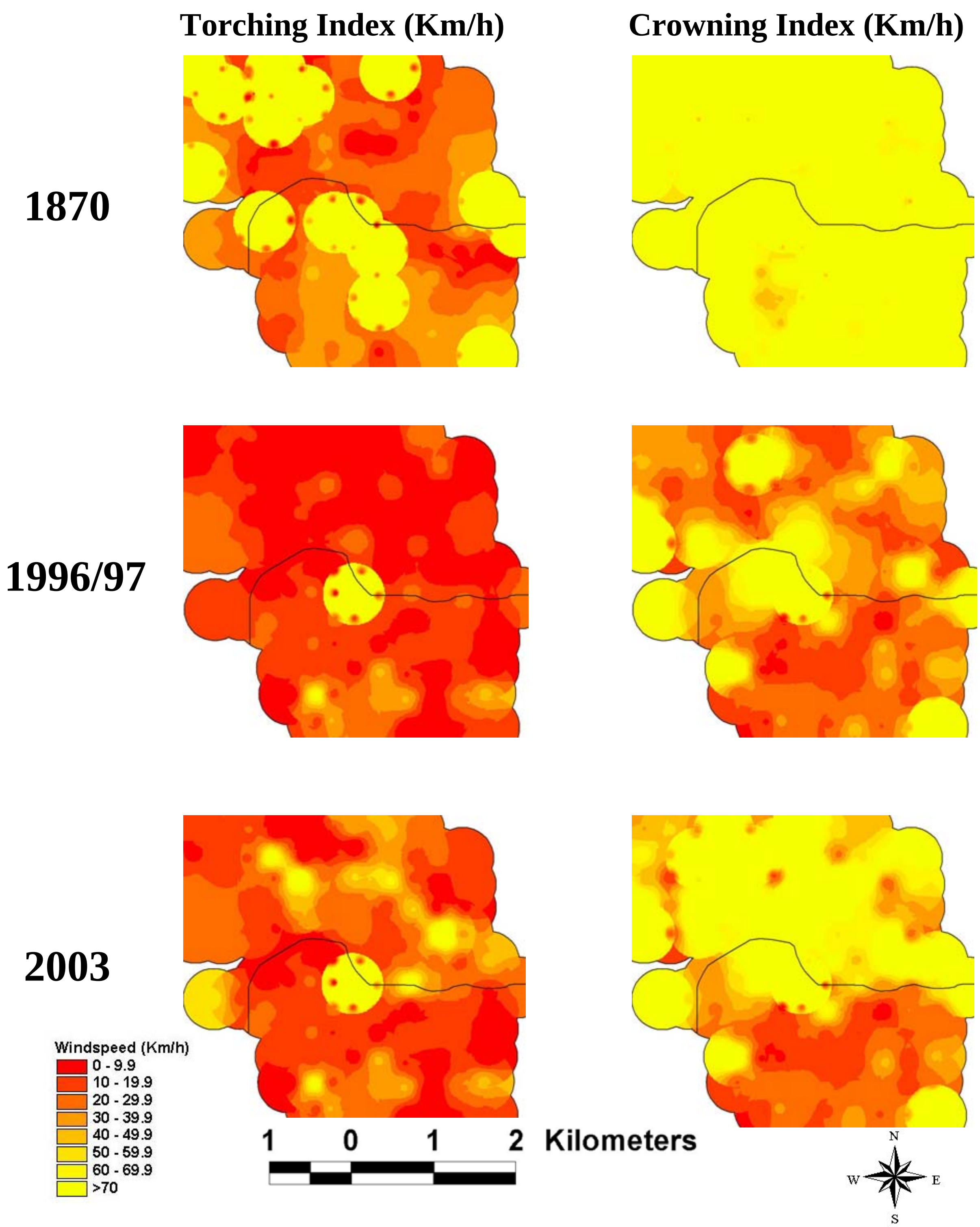

Figure 4.5 Torching and crowning indices throughout the Mt. Trumbull landscape were lowest in 1870, increased by 1996/97, and were reduced in the treated area (north half of the landscape) by 2003. Model inputs used were low quintile CBH, CBD using Cruz et al. (2003), and 80% foliar moisture content.

Chapter 5

Conclusions

Summary

The objective of this research was to determine if landscape-scale restoration treatments at Mt. Trumbull were implemented as intended and if they effectively restored historical forest structure conditions while allowing for the reintroduction of surface fire. This study provided the first detailed information regarding the implementation of landscape-scale ecological restoration treatments in a southwestern ponderosa pine ecosystem. Evaluation of treatment implementation is a vital component of the adaptive management process. It would be difficult to justify altering treatment prescriptions or continuing the current management approach if we were unsure whether the original prescriptions were followed. This study also assessed whether restoration treatments were a valid means of attaining the ultimate project goals set out by managers and researchers.

The success of treatment implementation was variable. Most of the area originally planned for restoration was treated in some manner by 2003; however, only 70% received the full planned treatment (thin and burn). Although pine density decreased significantly over the treated area, post-treatment levels were 111-256% above the projected density. Pine density exceeded the projected density by only 10-85% in areas that received the intended treatment (thin and burn). Despite contract amendments to terminate oak cutting, some oaks were still cut due to administrative lag. Eighty percent of the presettlement pines alive prior to treatment remained alive by 2003 in the treated area;

however, 4% of the presettlement oaks were cut and 10% died. One-third of large snags were lost, falling below the snag retention target, but new large snags were recruited, resulting in a net increase in snag density over the landscape. Implementation goals for large logs were achieved.

Restoration treatments were an effective means of attaining the overall project goal of restoring more open forest structure conditions while preserving the majority of the presettlement trees. Although density and basal area levels were more than double the 1870 values, 82% of the treated area was classified as having low stand density index (SDI) in 2003 compared to 44% before treatment. The diameter distribution of ponderosa pine, while still skewed compared to reconstructed distributions, was reduced in the proportion of smaller trees. Current regeneration densities were more than sufficient to sustain the current species composition and desired forest structure.

Restoration treatments were also effective at reducing canopy fuels and crown fire hazard. Canopy fuel load (CFL) and canopy bulk density (CBD) were both decreased substantially in the treated area, while slight increases occurred in the control. Canopy base height (CBH) was slightly higher in the treated area than in the control. Predicted outcomes were consistent between the two fire behavior models (FlamMap and Nexus): under extreme drought and wind conditions, crown fire hazard was reduced in the treated area. In contrast, the models show little change in active crown fire hazard in the control over the same time period. We conclude that the restoration treatments have successfully addressed the overall project goal of restoring forest structure and ecosystem processes within the historical range of natural variability and have reduced canopy fuels and crown fire hazard.

Management Implications

The Mt. Trumbull Ponderosa Pine Ecosystem Restoration Project serves as an excellent example of collaboration. In addition to developing a dedicated pursuit to restore the degraded Mt. Trumbull ecosystem, managers and researchers have followed through with a strong commitment to monitor ecological responses to restoration treatments. This study represents the first landscape-scale evaluation in the adaptive management process. Although restoration treatments were not implemented perfectly, the overall goal of re-establishing a sustainable and functioning ecosystem was achieved. After treatment, the restored area at Mt. Trumbull was structurally more heterogeneous and more similar to pre-1870 conditions than the untreated control. Although the treatments implemented were different than originally planned, the diversity of treatments created a variety of ecosystem conditions which will benefit a wide range of plants and animals as well as humans. From related finer-scale studies, there is reason to expect that these changes will result in improved ecosystem function (Covington et al. 1997, Kaye et al. 2005), increased vigor of old and young trees (Feeney et al. 1998, Stone et al. 1999, Skov et al. 2004), improved resistance to disturbance agents such as bark beetles (Wallin et al. 2004) and fire (Fulé et al. 2001, Chapter 4), sufficient regeneration (Bailey and Covington 2002), and increased productivity of herbaceous understory vegetation (Covington et al. 1997, Laughlin et al., in press, Moore et al., in press). However, treatments have also resulted in the loss of some old trees from prescribed fire activities (Fulé et al. 2002, Jerman et al. 2004) and the spread of the invasive exotic *Bromus tectorum* (C. McGlone, pers. comm.). Wildlife effects documented at Mt. Trumbull have been mixed to date, with beneficial and negative aspects depending on the animal species and scale of study (Germaine and Germaine 2002, Battin 2003, Germaine et al. 2004, Waltz and Covington 2004).

Evolution of treatments over time is common in broad-scale, extended management projects, but it is rare to have access to detailed data from permanent plots to assess changes. Lessons from this study include, first, that ongoing monitoring can be very helpful in identifying problems. We determined relatively early that cutting of oaks and heat effects of burning were issues of concern. Second, even after identifying issues there can be an administrative lag until changes take effect. Oaks in thinning contract areas, for instance, were thinned even after the decision was made to stop. Third, our data have identified new areas on which to focus attention in restoration treatments. Old ponderosa pines were largely uninjured, but old oak trees had a high rate of mortality. Future projects should maintain pine protection while addressing oak survival more explicitly. Fourth, monitoring offers quantitative data on which to rest decisions about future treatments. Managers planning future restoration treatments can draw upon these lessons for developing new treatments and modifying existing prescriptions. This study underscores the importance of including a comprehensive, long-term monitoring plan in proposed resource management projects so that implementation and effectiveness of treatments can be evaluated. Such extensive monitoring may not be feasible due to resource limitations. Therefore, managers should make every possible effort to incorporate some level of monitoring into projects so that prescriptions can be evaluated and altered if necessary.

The Mt. Trumbull ecosystem will never be “fireproofed.” Some level of crown fire will likely occur in the future, particularly in untreated areas. Even within treated areas, passive crown fire may occur, especially during dry years. However, the overall management objective of reducing canopy fuels and crown fire hazard was achieved in treated areas. Maintenance of the surface fire regime will be vital to retaining open forest conditions and relatively low crown fire hazard into the future. The Mt. Trumbull area contains additional dense ponderosa pine forests. If these areas remain untreated, they will remain vulnerable to stand-replacing crown fires which could cause large patches with high tree mortality and could potentially limit conifer regeneration (Barclay et al. 2004). Severe fires may even result in ecosystem conversion to shrubfields or grasslands (Savage and Mast 2005, Strom 2005). The Mt. Trumbull ponderosa pine ecosystem is not yet “restored”. However, restoration treatments have been successful at reducing crown fire hazard and creating more sustainable and dynamic forest conditions.

Future Research

The Mt. Trumbull Ponderosa Pine Restoration Project has provided myriad research opportunities to date and further opportunities will likely continue if adequate funding is maintained and if ecosystem restoration remains a priority. Because the remainder of the planned restoration treatments will presumably be implemented within the next few years, subsequent landscape-scale assessments of restoration implementation and effectiveness and potential fire behavior should include data from the entire plot grid and should occur approximately once per decade. Inventories of ponderosa pine seedlings should be conducted once every five years until verification of adequate regeneration of this species is documented. It is important that monitoring continues so the adaptive management process functions as designed. Finally, due to its scientific value, the control should remain untreated indefinitely.

The BLM currently manages two wilderness areas at Mt. Trumbull. Research plots were installed in the Mt. Logan Wilderness (Waltz and Fulé 1998) and the Mt. Trumbull Wilderness (Heinlein et al. 1999) in the late 1990's to inventory contemporary forest conditions. Tree condition should be remeasured on these plots so presettlement tree mortality can be assessed. In addition, potential fire behavior should be assessed. The results from these proposed assessments and the results from the current study can be used in the environmental assessments of these two wilderness areas to evaluate the level of ecological degradation and demonstrate the potential benefits of restoration treatments. Additional studies should be initiated to develop and test a range of treatment alternatives for these ecologically important areas.

Aerial photographs from 1940, 1992, 2003, and other years exist for the Mt. Trumbull area. A change-over-time study should be conducted using these remotely sensed data to supplement the on-the-ground data used in the current study. Landscape metrics such as mean patch size, area to edge ratio, and connectivity could be used to describe the Mt. Trumbull landscape in 1940, 1992 (pre-treatment), and 2003 (post-treatment) and would be particularly beneficial to wildlife researchers.

The efforts put forth by the BLM have resulted in the implementation of landscape-scale restoration treatments at Mt. Trumbull and have provided researchers with a "living laboratory" for ponderosa pine restoration research. Results from continued monitoring of existing studies and new research at Mt. Trumbull will provide managers and researchers opportunities to address future resource management challenges.

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