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EFFECTIVENESS OF STAND-SCALE FOREST RESTORATION,  
SISKIYOU MOUNTAINS, OREGON

By

Jay Charles Lininger

Professional Paper

presented in partial fulfillment of the requirements  
for the degree of

Master of Science  
in Environmental Studies

The University of Montana  
Missoula, MT

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## ABSTRACT

This case study documents the effectiveness of stand-scale forest restoration activities undertaken in the “Penny Stew” project on federal land in the central Siskiyou Mountains of southwest Oregon. Its dual purpose is to 1) define reference conditions and build a site-specific case for restoration through multi-scale analysis of historical ecology (Chapter III) and 2) demonstrate streamlined monitoring protocols that practitioners can use to adapt restoration practices over time (Chapter IV). Discussion of methodological efficacy in the monitoring report offers implications for FEAT-FIREMON Integration (FFI), a fire effects monitoring and data analysis software package currently under development for use by wildland fire managers. Finally, this analysis synthesizes findings from reference analysis and monitoring results to assess the degree to which the Penny Stew project meets restoration objectives, and it recommends further action including application of management-ignited prescribed fire and sustained monitoring in light of that synthesis (Chapter V).

## ACKNOWLEDGEMENTS

I wish to extend deep thanks to the people who helped me conceive, design, execute, and critique this project. I am especially grateful to Jason Clark, Tim Holloway, Luke Ruediger, and Joseph Vaile for their invaluable field assistance including plot layout, data collection, botanical identification, and photo documentation. Marko Bey, Oshana Catrinides, and Justin Cullumbine inspired this project and arranged financial and logistical support. The Doris A. Duke Foundation also provided a stipend that funded one season of field work. Paul Alaback, Li Brannfors, John Carratti, Alison Forrestel, Richard Hart, Bob Keane, Duncan Lutes, Wende Rehlaender, and Ron Wakimoto instructed me in the field methods and data management tactics used in this study. Len Broberg, Carl Fiedler, Frank Lake, and Dennis Odion supplied helpful reviews and encouraged me to broaden my thinking about relevant issues and literature. Timothy Ingalsbee and Marty Main offered continuous encouragement, and Liza Tran does not realize her profound impact on my ability to see this project through to completion. Deep thanks to everyone.

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## I. INTRODUCTION

Ecosystem management in the western United States concentrates extensively on restoration of fire-adapted forests. Federal Wildland Fire Policy (USDA/USDI 2001) recognizes that fire suppression and logging in western forests have caused widespread ecological deterioration and corresponding losses of biological diversity (“biodiversity”). In many places, biodiversity conservation may depend on restoration of wildland vegetation structure and active function of natural fire disturbance processes (Hardy and Arno 1996). As a result, lands that exhibit significant departure from historical fire regimes now receive priority for active management including mechanical fuel reduction and wildland fire use (USGAO 2005).

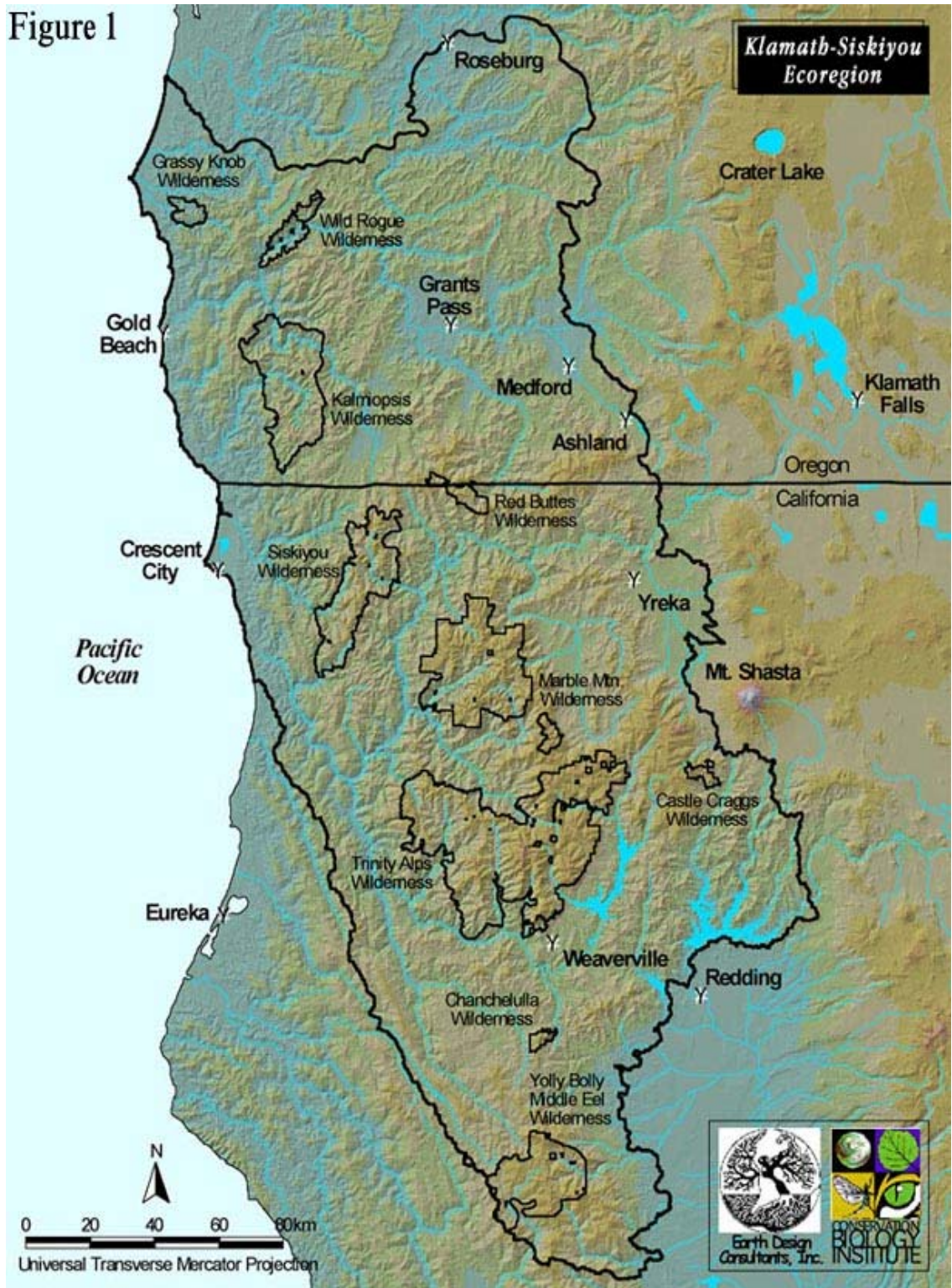
Substantial research demonstrates a need for active restoration in dry ponderosa pine (*Pinus ponderosa*) forests dominated by frequent, low-severity fire regimes in the American Southwest and Inland West (Allen et al. 2002, Hann et al. 1997, Moore et al. 1999, Noss et al. 2006). Ecosystem managers commonly cite this research to justify intervention in other fire-adapted forest types (e.g., USDA 2005, USDI 2005). Fire regimes, however, vary over time and space (Agee 1993). For example, the seasonal timing of fire disturbance strongly influences plant and ecosystem responses to burning, and fire seasonality varies along latitudinal gradients (Brown 2000a). Mixed conifer forests in the Sierra Nevada of California mainly experience fire in the summer through autumn when plant growth is dormant (Caprio and Swetnam 1995), whereas similarly composed forests in the Sierra San Pedro Martir of Baja, Mexico, often burn in the spring and early summer growth seasons (Stephens et al. 2003). This latitudinal gradient of fire



seasonality is related to geographic variation in the onset and length of drought periods, which can influence the duration of time when fuels are dry enough to burn each year (Taylor and Skinner 2003). As a result, fire-induced vegetation responses and succession patterns are regionally distinct (Odion et al. 2004).

Mixed evergreen forests of the Klamath-Siskiyou (KS) region (Figure I-1) historically depend on wildland fire to shape their structure, composition, nutrient flows, hydrology, habitat and other ecosystem qualities (Atzet and Martin 1991). Indigenous organisms evolved with fire to varying degrees, and the life history of many endemic plants requires fire to persist (Martin 1997). Fine-scale variations in geology, climate, topography and vegetation influence regional fire regimes and landscape pattern (Taylor and Skinner 2003). Indeed, the unusual environmental variability and “pyrodiversity” of the KS region supports temperate biodiversity of globally outstanding conservation importance (DellaSala et al. 1999, Martin and Sapsis 1992).

The highly complex landscape fire regime of KS mixed evergreen forests precludes generalization about fire history and effects (Agee 1993). Ecosystem managers routinely use coarse-scale fire regime condition classification of vegetation, fuel and disturbance to index landscape departure from historical fire regimes and identify lands at-risk of uncharacteristically severe fires that may impair ecosystem function (Hann and Bunnell 2001). Such assessments characterize most KS forests as “condition class 3,” or severely altered from historical conditions (Schmidt et al. 2002). However, fire regime condition class poorly predicts actual wildland fire effects (Odion and Hanson 2006), and researchers demand convincing evidence of ecosystem departure from adapted disturbance regimes before ecologically unprecedented restoration interventions are



**Figure I-1.** Klamath-Siskiyou region of northwest California and southwest Oregon.  
 Source: Strittholt et al. (1999).

undertaken (Gutsell et al. 2001).

Given that ecosystem management based on natural disturbance regimes “will always be somewhat uncertain” (Landres et al. 1999), conservation biologists urge precaution in decision-making about ecological restoration when systems thought to be degraded are not well understood (Noss and Cooperrider 1994). The precautionary principle counsels against actions that cannot be reversed later if the decision is wrong (Meffe and Carroll 1997). In this view, restoration should target areas most likely to benefit from active intervention (Brown et al. 2004). Need for restoration depends on ecological scale, disturbance history, vegetation characteristics and current conditions (Lindenmayer and Franklin 2002).

Large areas of the KS region remain little disturbed by human management and closely resemble conditions in which indigenous life evolved (Staus et al. 2003). Places retaining high degrees of ecological integrity generally host few if any roads (Strittholt et al. 1999). Those places function as reservoirs of biodiversity where passive restoration (i.e., halting or foregoing activities that may cause ecological damage) and active use of wildland fire for resource benefits may offer the most ecologically sensible management approaches over time (DellaSala and Frost 2001). However, legitimate needs for more active restoration of KS forests often exist in areas with more substantial road densities, particularly at lower elevations where intensive human use history overlaps drier forest types that are most likely to have experienced functional alteration due to cumulative effects of logging and fire exclusion (DellaSala et al. 2004).

Ecologists stress the importance of defining locally specific reference conditions to justify restoration goals and outcomes (White and Walker 1997). Descriptions of

natural variation in ecosystems derived from historical ecology and their application as reference conditions to land management are matters of controversy (Swetnam et al. 1999). However, it is generally accepted that understanding historical ecosystem dynamics, structures and functions can provide useful information to guide restoration efforts (Society for Ecological Restoration International Science & Policy Working Group [SER] 2004).

The inherent complexity and dynamism of ecological systems render impossible accurate prediction of all consequences of restoration activities. Therefore, such projects initially should be confined to small spatial scales and accompanied by monitoring and evaluation sufficient to inform adaptive management (DellaSala et al. 2004). Monitoring facilitates impact assessment and tactical adaptation if treatments produce unintended or inadequate results (Lee 1993). Monitoring also empowers restoration practitioners to demonstrate contract compliance, educate stakeholders and elevate restoration discourse above “faith-based forestry” (Bey 2005). Funding, complexity, training and commitment can pose formidable barriers to reliable effectiveness monitoring of ecological restoration (Elzinga et al. 1998). Consequently, there exists a need for streamlined monitoring protocols that simplify and improve efficiency of the task without compromising defensibility.

In 2004, the Medford District of the U.S. Bureau of Land Management (BLM) contracted the non-profit Lomakatsi Restoration Project (LRP) to undertake mechanical thinning and burning treatments to alter forest structure and composition as well as reduce hazardous fuel and facilitate stand-scale ecological restoration in the “Penny Stew” project. Penny Stew is a small and non-controversial element of a broader forest

management project (USDI 1999) that drew a successful legal challenge from conservationists concerned about the removal of old, large and fire-resilient trees in a timber sale. It offers an important opportunity for ecosystem managers and practitioners to demonstrate restoration forestry practices, and in doing so, promote social consensus on active restoration of fire-adapted forests on federal public land. As part of this demonstration effort, the LRP seeks to define reference conditions for restoration and monitor the effectiveness of its treatments.

This case study documents the effectiveness of stand-scale forest restoration activities undertaken in the Penny Stew project. Its dual purposes are to (1) define reference conditions and build a site-specific case for restoration through multi-scale analysis of historical ecology (Chapter III), and (2) demonstrate streamlined monitoring protocols that practitioners can use to adapt restoration practices over time (Chapter IV). It synthesizes findings from reference analysis and monitoring results to determine the degree to which the Penny Stew project meets restoration objectives, and it recommends further action and monitoring in light of that synthesis (Chapter V).

## II. STUDY AREA

The study area is located in the central Siskiyou Mountains of southwest Oregon. The Siskiyou Mountains are part of the broader Klamath Mountains Province (Fenneman 1931), also called the Klamath-Siskiyou (KS) region (DellaSala et al. 1999), in the northern portion of the California botanical province (Jepson 1966) (Figures I-1, II-1). The KS region holds unusual ecological interest with its old and geologically complex mountains and diverse floristic patterns in relation to steep climatic, edaphic and topographical gradients (Coleman and Kruckeberg 1999). Its central location is transitional to the Great Basin, Oregon and California Coast Ranges, Cascades Range, Sierra Nevada and California Central Valley, sharing vegetative influences from each (Whittaker 1961). Partly as a result, it features the richest assemblage of vascular plant species of any geologic province in western North America (Wagner 1997). Bedrock geology and soil chemistry primarily influence the spatial distribution of vegetation patches at a landscape scale, while soil moisture and terrain exert significant secondary influences (Whittaker 1960). Forest and woodland vegetation occupy mountainous topography associated with nutrient-rich rock forms and span abrupt gradients of elevation, slope, aspect and moisture (DellaSala et al. 1999).

The KS region consists of four belts of rock that formerly were oceanic crust and island arcs. Around 150 to 350 million years before present (BP) the oceanic material scraped off and welded to the western edge of the North American continent (Coleman and Kruckeberg 1999). Two of those rock belts are present in the Applegate River watershed (Figure II-1). About 150 million years BP, the older of the two thrust-faulted



**Figure II-1.** Location of the Applegate River watershed and the Klamath Mountains Province relative to other physiographic provinces in the Pacific Northwest. Source: USDA/USDI (1995).

over the younger one for a distance of approximately 100 kilometers (km). Later, an uplift of up to 7,000 vertical meters (m) centered on Condrey Mountain occurred (Prchal n.d.), which accounts for the steepness of local mountain terrain. Glaciation affected parts of the high Siskiyou Mountains above 1,200 m from about three million years BP until about 11,000 years ago, but there is little evidence of glacial activity beyond that. Most of the KS region therefore functioned for millennia as a land bridge for animal and plant migrations between the Great Plains and the Pacific Coast (Whittaker 1961).

The study site is located on the slopes of Pennington Mountain, a low-elevation foothill of the central Siskiyous that divides Williams Creek from the Applegate River canyon (Figure II-2). Pennington Mountain lay in a partial rain shadow created by Grayback Mountain on the Siskiyou Crest. Seasonal moisture patterns currently feature cool, wet winters and hot, dry summers. Average annual precipitation is approximately 90 centimeters (cm) with the driest time occurring in July, which annually averages <1 cm of rainfall. Snow occasionally graces the area, but sites lower than 670 m above sea level usually do not receive significant snowfall (Roether et al. 2000, USDI 1996).

The specific location of the study area is in section 3 of Township 38 South, Range 5 West of the Willamette Meridian. It is divided into two distinct land units. Unit 1 to the south of Pennington Mountain comprises approximately 16.2 hectares (ha) bound to the north by Water Gap Creek and to the south by Pennington Creek, both of which flow into Williams Creek, an Applegate River tributary (Figure II-3). Its elevation spans from 457 to 515 m above sea level. Abegg loam and Manita loam soils underlay the unit (USDA 1983). Each soil type is deep, well-drained and generally occurs on gentle (2-7%) slopes on high stream terraces and foothills adjacent to river valleys.



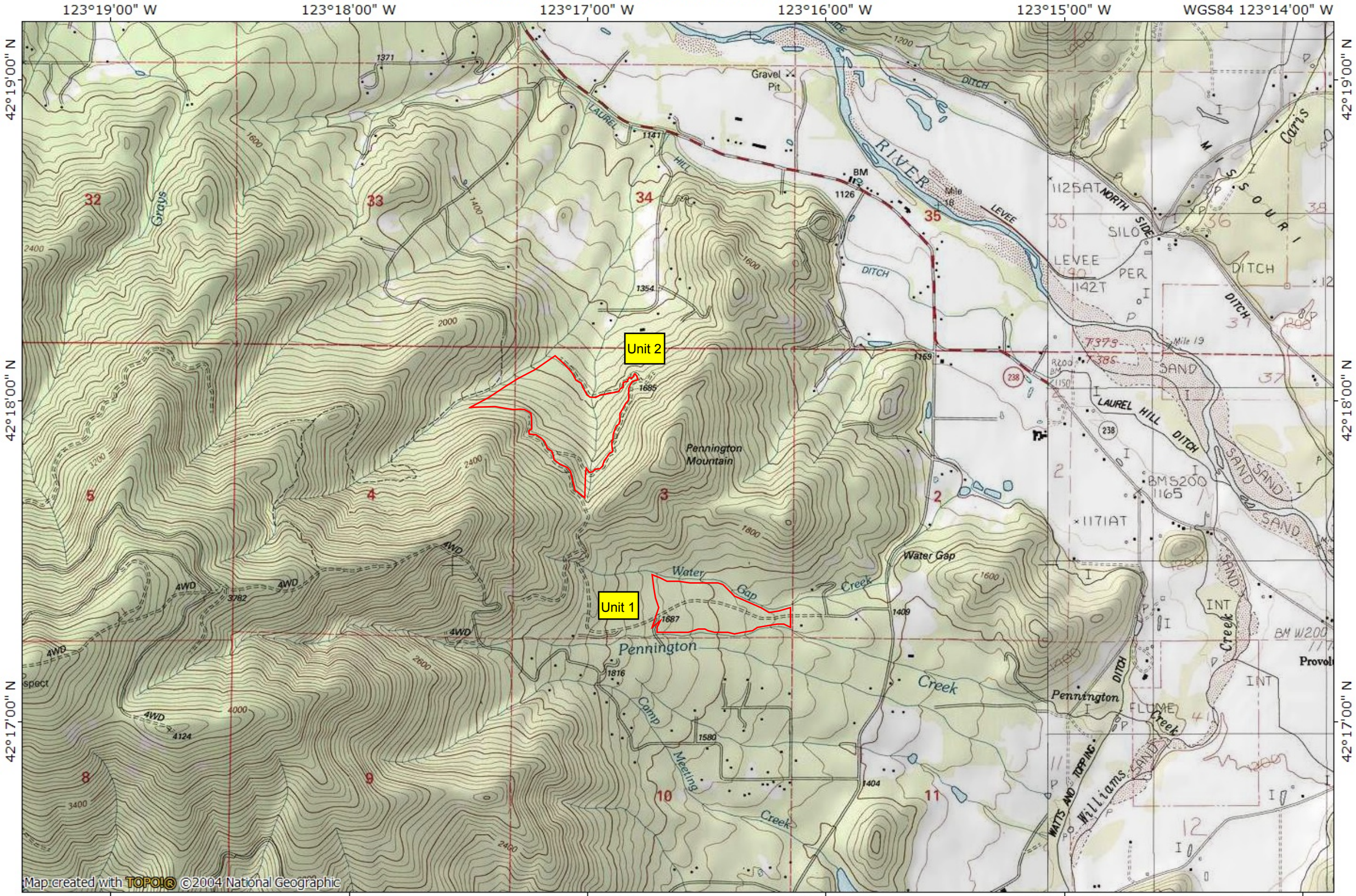


**Figure II-2.** Study area location in the Applegate River watershed of the central Siskiyou Mountains in southwest Oregon. Source: USDA/USDI (1994a).

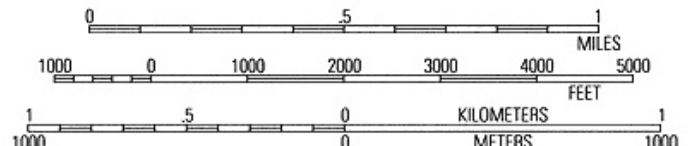
Unit 1 hosts xeric valley woodland and forest vegetation typical of low elevation sites in the western Applegate River watershed (Atzet et al. 1996, Hickman 1995).

Unit 2 encompasses ~25.5 ha draining unnamed tributaries to the Applegate River on the north face of Pennington Mountain (Figure II-3). Its elevation ranges from 472 to 610 m above sea level. Manita loam soil characteristic of steep (35-50%) foothill slopes underlays its southwest corner, though most of Unit 2 features a Cornutt-Dubakella clay soil complex on rolling (7-20%) slopes and alluvial fans. The latter derives from mixed

parent materials including ultramafic rock with high magnesium and low calcium contents. Cornutt soil is less influenced by ultramafics than Dubakella, and root depths in Cornutt (100-150 cm) exceed those in Dubakella (50-100 cm) (USDA 1983). The former hosts relatively productive mixed evergreen forest dominated by mature Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) and the latter supports a patch of California black oak (*Quercus kelloggii*) woodland near the unit center.



Map created with **TOPOIC** ©2004 National Geographic



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### III. REFERENCE ANALYSIS

#### **Introduction**

Reference condition analysis facilitates goal setting in ecological restoration (Swetnam et al. 1999). Practitioners need to define reference conditions because they help to (1) determine what factors cause ecological degradation, (2) identify what needs to be done to restore an ecosystem, and (3) inform criteria that measure success of restoration treatments (SER 2004). Egan and Howell (2001) suggest historical ecology as a foundation for reference description recognizing that no single technique is adequate due to shortcomings inherent to each. This is self-evident considering the scale dependence of observed variations in ecosystem pattern and function (White and Walker 1997). For example, paleoecological research on climate interactions with flora and disturbance offers temporal depth but operates at very coarse scales. Similarly, tree ring studies can demonstrate point-scale fire effects on vegetation over shorter timeframes but often lack modern calibration and can mislead investigators about historical fire regimes (Baker and Ehle 2003).

Understanding the spatial and temporal contexts in which ecosystems function is critical to framing a coherent restoration strategy (Landres et al. 1999). For this reason, Noss (1985) suggests using a multi-scale approach to defining reference conditions. Following that suggestion, Morgan and others (1994:90) advocate use of “historic range of variability” (HRV) to describe broad-scale and long-term ecological dynamics:

[HRV] can serve as a useful tool for understanding the causes and consequences of change in ecosystem characteristics over time. It provides a context for interpreting natural processes, especially disturbance, and it allows variability in patterns and processes to be understood in terms of a dynamic

system. Study of past ecosystem behavior can provide the framework for understanding the structure and behavior of contemporary ecosystems, and is the basis for predicting future conditions.

HRV is particularly relevant where negative effects on ecosystem function and biodiversity accompany observable changes in ecological conditions over time. For example, fire exclusion and logging in some ponderosa pine forests of the Inland West may have altered ecological function such that existing systems are vulnerable to catastrophic loss (Hann et al. 1997). However, this idea is controversial because historical fire regimes are poorly understood, particularly where fire disturbance patterns vary in extent, timing, intensity and biological effects (Baker et al. 2006, Veblen 2003). Obvious departures from a clearly defined HRV may justify active restoration (Arno and Fiedler 2005), but in some cases, passive restoration including cessation of activities that degrade ecosystems (e.g., fire exclusion) may be sufficient (DellaSala et al. 2004).

Ecosystem managers in the KS region commonly use forest density, structure and composition that existed pre-1900 as reference conditions for restoration of fire-adapted forests (e.g., USDA 2003, USDI 2005). However, reliance upon this temporally limited and structurally focused set of reference conditions overlooks shifts in climate and fire regimes that occurred in the 20<sup>th</sup> century (Running 2006). Indeed, climate change in the Pacific Northwest may preclude forest ecosystems from sustaining conditions that existed concurrent with European settlement (Whitlock et al. 2003). Therefore, any discrete historical condition is not as useful to restoration as “a range of forest conditions that approximates those historically adapted to the fire regime,” not merely because they are historical, but because they are self-perpetuating and resilient to disturbance (Arno and Fiedler 2005:38-39). Understanding how forests respond to climatic changes over long

timescales may offer clues about their resiliency and adaptability (Millar and Woolfenden 1999).

Additionally, in many areas, human activities exerted significant influences on historical ecosystems. People lived in southwest Oregon at least 7,000 years before European contact and probably much longer (Winthrop 1993). Another way to approach description of reference conditions in the study area is to consider how human activities may have affected ecosystem structure and function, and how those activities might have responded to climate dynamics. Williams (2000) argues that ecological crisis in western forests is rooted in the cessation of indigenous fire use practices. Accounts of aboriginal burning are found in old notes, journals and the oral tradition. Such qualitative and anecdotal sources are not readily accepted by scientists whose training traditionally is limited to interpretation of quantitative data (Kimmerer and Lake 2001). Nevertheless, Anderson (1997) argues for better rapprochement between the social, historical and biological sciences in learning about wild plant production and other ecological functions that undeniably were central to human livelihoods in the past.

This chapter synthesizes a multi-disciplinary investigation of historical vegetation surveys, fire histories, paleoecological studies of climate, anthropological and ethnographic data, and traditional ecological knowledge to formulate a qualitative description of reference conditions for the study area. It strives to account for spatial and temporal variations that have influenced forest ecosystem structure and function while offering a practical outline for stand-scale restoration activities. The cumulative weight of information considered permits cautious inference of reference conditions based on known plant characteristics, disturbance patterns, and human cultural practices.

## **Method**

### *Documentary research*

We consulted watershed-scale assessments of vegetation and landscape condition to determine the study area's ecological context, and we attempted to locate nearby reference sites featuring high ecological integrity and lack of management effects. We also reviewed General Land Office (GLO) maps and survey data to ascertain settlement era conditions in and around study site. We then examined regional fire ecology studies and ethnographic research to ascertain disturbance ecology and human use patterns. Finally, we considered paleoecological research on regional climate to inform our characterization of ecological variation and shifting human uses of natural resources over time.

### *Site investigation*

We sampled forest structure and vegetative composition in six randomly located plots located inside the study area using the protocols described in Chapter IV. We recorded the density and relative cover of trees, shrubs and herbs, and entered this data into customized database software (JFiremon). We selected the most abundant plant species and researched their habitat preferences and disturbance adaptations in the Fire Effects Information System (<http://www.fs.fed.us/database/feis/>). We also consulted Atzet and others (1996) to classify plant associations based on site-specific vegetation cover and frequency data, and in turn used this information to verify the applicability of disturbance ecology research to the study area.

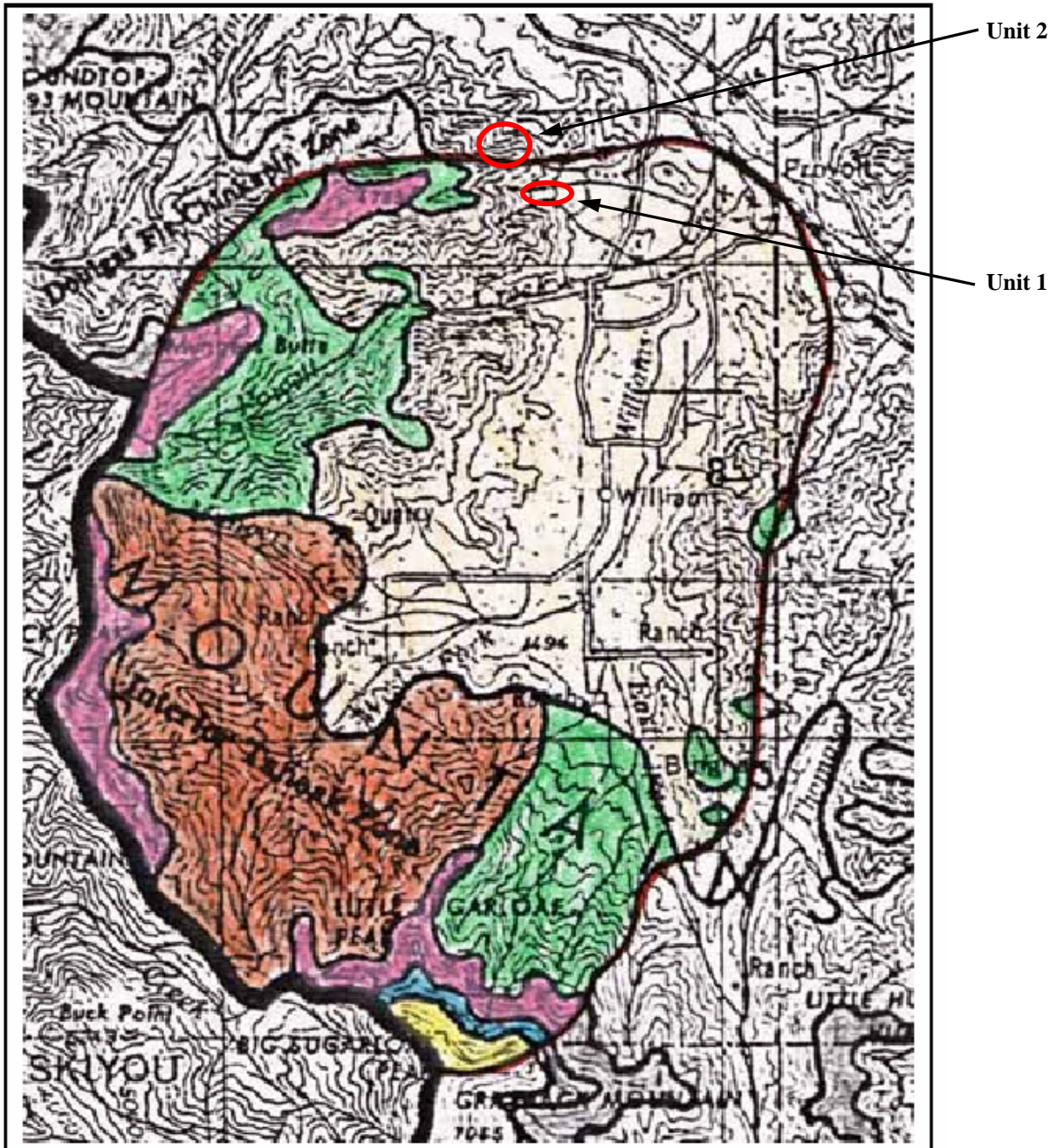
## Results and discussion

### *Plant composition*

The study area straddles the Interior Valley Zone and Mixed Conifer Zone of the Applegate River watershed described by Hickman (1995) (Figure III-1). The former includes low-elevation river valleys as well as portions of surrounding foothills. Several distinct vegetation communities occur in the Interior Valley Zone, including grasslands, chaparral and oak-conifer woodlands (Hickman 1995). Common plants in this zone historically were important to indigenous people, furnishing dietary staples such as acorns and camas, a variety of seeds and abundant forage for game (Winthrop 1993). In contrast, the Mixed Conifer Zone is transitional between the dry oak-conifer woodland communities of lower elevations and moist coniferous forest zones that occur at higher elevations (Figure III-1). Mixed conifer forest commonly occurs on northerly aspects of foothills above the Applegate River (Roether et al. 2000, USDI 2000).

Table III-1 displays common native plants in the study area. The southern half of the study area (Unit 1) hosts an oak-conifer woodland characteristic of hot, dry foothill sites in the Interior Valley Zone. Plot data indicate that vascular species richness is 25 species per plot, and shrub and herb cover is very low (<5%). We observed the existence of flora representative of the Douglas-fir/California black oak/poison oak (*Pseudotsuga menziesii*/*Quercus kelloggii*/*Toxicodendron diversiloba*) plant association described by Atzet and others (1996). This is the warmest and driest of all plant communities in the Douglas-fir series of southwest Oregon. The overstory canopy is dominated by sugar pine (*Pinus lambertiana*), ponderosa pine (*P. ponderosa*) and Douglas-fir. Black oak and Pacific madrone (*Arbutus menziesii*) occupy the understory along with Douglas-fir.





Vegetation Zone Legend	
<b>Low elevation zones</b>	<b>High elevation zones</b>
Dry Oak Conifer Zone: white	Cool Mixed Conifer Zone: pink
Interior Tanoak Zone: orange	White Fir Zone: blue
Moist Mixed Conifer Zone: green	Shasta Fir Zone: yellow
Douglas Fir-Chinkapin Zone: green	

Figure III-1. Vegetation zones of the Williams Creek watershed. Source: Hickman (1995).

**Table III-1.** Common native plants in the study area

Species	Form	Common name	Dry oak-conifer	Mixed conifer
<i>Abies concolor</i>	t	white fir	p	p
<i>Acer macrophyllum</i>	t	big-leaf maple	-	p
<i>Arbutus menziesii</i>	t	Pacific madrone	M	m
<i>Arctostaphylos viscida</i>	s	whiteleaf manzanita	p	p
<i>Alnus rubra</i>	t	red alder	-	p
<i>Amalancier alnifolia</i>	s	western serviceberry	-	p
<i>Berberis piperiana</i>	s	Piper's Oregon grape	p	p
<i>Brodiaea elegans</i>	f	harvest brodiaea	p	p
<i>Bromus carinatus</i>	g	California brome	p	p
<i>Calocedrus decurrens</i>	t	incense cedar	p	p
<i>Calochortus tolmiei</i>	f	pussy ears	p	p
<i>Calypso bulbosa</i>	f	fairy slipper	-	p
<i>Ceanothus integerrimus</i>	s	deer brush	p	p
<i>Claytonia lanceolata</i>	f	western spring beauty	p	p
<i>Cornus nuttallii</i>	t	mountain dogwood	-	p
<i>Dodecatheon hendersonii</i>	f	shooting star	p	p
<i>Elymus glaucus</i>	g	blue wildrye	p	-
<i>Erythronium hendersonii</i>	f	fawn lilly	p	p
<i>Festuca californica</i>	g	California fescue	p	m
<i>Goodyera oblongifolia</i>	f	rattlesnake plantain	p	p
<i>Hieracium albiflorum</i>	f	white-leaved hawkweed	p	p
<i>Holodiscus discolor</i>	s	creambrush oceanspray	p	m
<i>Iris tenax</i>	f	slender-tubed iris	p	p
<i>Lonicera hispidula</i>	s	hairy honeysuckle	m	m
<i>Madia madioides</i>	f	woodland tarweed	p	p
<i>Melica californica</i>	g	California melic	p	p
<i>Nasella pulchra</i>	g	purple needlegrass	-	p
<i>Osmorhiza chilensis</i>	f	sweet cicily	p	p
<i>Quercus chrysolepis</i>	t	canyon live oak	p	m
<i>Quercus kelloggii</i>	t	California black oak	M	m
<i>Pedicularis densiflora</i>	f	lousewort	p	-
<i>Perideridia howellii</i>	f	yampah	p	p
<i>Pinus ponderosa</i>	t	ponderosa pine	M	m
<i>Pinus lambertiana</i>	t	sugar pine	M	m
<i>Pseudotsuga menziesii</i> var. <i>menziesii</i>	t	coastal Douglas fir	m	M
<i>Pyrola picta</i>	f	white-veined wintergreen	p	p
<i>Rosa gymnocarpa</i>	s	wood rose	m	p
<i>Rubus ursinus</i>	s	California blackberry	p	p
<i>Sanicula crassicaulis</i>	f	sanicula	p	p
<i>Symphoricarpus albus</i>	s	creeping snowberry	p	m
<i>Taxus brevifolia</i>	t	Pacific yew	-	p
<i>Toxicodendron diversiloba</i>	s	poison oak	M	M
<i>Tridentalis latifolia</i>	f	starflower	p	p
<i>Vicia californica</i>	f	vetch	p	p

f, forb. g, grass. s, shrub. t, tree.

M, major. m, minor. p, present.

Poison oak, hairy honeysuckle (*Lonicera hispidula*), deer brush (*Ceanothus integerrimus*) and whiteleaf manzanita (*Arctostaphylos viscida*) are common shrubs. The presence of manzanita in Unit 1 is noteworthy because it is representative of the northernmost extension of the California chaparral formation (Jepson 1966). In this portion of its range, chaparral frequently associates with oak, Douglas-fir and pine, and its presence indicates xeric climatic conditions. Moreover, chaparral does not exist in the absence of an active fire regime (Detling 1961).

The north half of the study area in Unit 2 supports mixed conifer and hardwood forest expressive of the Mixed Conifer Zone (Hickman 1995). Plot data suggest that vascular species richness is 32 species per plot, and shrub and herb cover is somewhat greater (5-10%) than in Unit 1. Vegetation in Unit 2 is representative of a Douglas-fir/ponderosa pine/poison oak association described by Atzet and others (1996). Douglas-fir dominates the overstory, and ponderosa pine and sugar pine also are common. Hardwoods including canyon live oak (*Quercus chrysolepis*), black oak and madrone frequently occur in the understory. Common shrubs including poison oak, deer brush, hairy honeysuckle, creeping snowberry (*Symphoricarpos albus*), western serviceberry (*Amalancier alnifolia*), creambrush oceanspray (*Holodiscus discolor*) and Piper's Oregongrape (*Berberis nervosa*) all indicate low soil moisture. However, the presence of mountain dogwood (*Cornus nuttalli*), big leaf maple (*Acer macrophyllum*), red alder (*Alnus rubra*) and Pacific yew (*Taxus brevifolia*) in close proximity to two ephemeral draws demonstrate the existence of mesic riparian microclimates in portions of the unit.

Plant composition of the study area invites application of Whittaker's (1960:312) description of a "mixed forest with an upper tree stratum of needle-leaved evergreen or

coniferous and a lower tree stratum of broad-leaved evergreen or sclerophyllous species,” which forms a unique botanical formation in the Siskiyou Mountains. Mixed evergreen forest commonly features various hardwood trees and chaparral shrubs intermingling with conifers on dry sites (Sawyer et al. 1977). The mixed evergreen formation occupies an elevation gradient ranging from the foothills above interior valleys and transitions at higher elevations to montane true fir forests where hardwoods are rare (Whittaker 1960). Douglas-fir is ubiquitous in mixed evergreen forest, occupying all aspects and slope positions. Pines most frequently occur on south aspects and at upper slope positions (Franklin and Dyrness 1988).

#### *Disturbance ecology*

Fire profoundly influences the structure, composition, soil properties, nutrient cycles, wildlife habitat and other qualities of mixed evergreen forests in the Siskiyou Mountains (Atzet and Martin 1991). Whittaker (1960:307) observed that mixed evergreen forests

may be regarded ... as a fire-adapted vegetation of a summer-dry climate, in which fires of varying frequencies and intensities and varying sources – white man, Amerind, and lightning – have for a very long time been part of its environment. If the term “climax” is to be applied in such circumstances, it seems supposititious to apply it to the non-existent vegetation which might develop after centuries of complete fire protection. The climax may better be regarded as that reasonably stable and self-maintaining vegetation which exists in this area, in adaptation to fires and other factors of environment. It may be understood in this case that the climax, or fire-climax, condition embodies a degree of population instability and irregularity resulting from fires affecting different areas in a patch-wise fashion at irregular intervals.

Fire is an intrinsic ecological process in mixed evergreen communities affecting plant evolution and succession patterns at variable spatial and temporal scales (Frost and Sweeney 2000). The plant association concept discussed above therefore offers limited

utility because it flows from the potential natural vegetation model of Kuchler (1964), which assumes that community succession is a unidirectional process leading to a “climax” steady-state condition in the absence of disturbance. In fact, plants respond individually to environmental conditions, creating irregular gradients in floristic communities over space and time (Gleason 1939, Whittaker 1975).

Sources of spatial variation in fire extent include topographic breaks such as rock fields and moist riparian habitats that can inhibit fire spread when weather is not extreme, creating barriers that filter its movement across the landscape (Taylor and Skinner 2003). As flames encounter different terrain, weather and fuel conditions, the intensity of heat energy output and the severity of effects on vegetation and soil fluctuate in complex patterns (Miller 2000). The steep topography of the Siskiyou Mountains can facilitate wind-driven convection currents that drive radiant heat upward and bring flames nearer to adjacent, unburned vegetation, thus pre-heating fuels and amplifying fire intensity as it moves upslope. Partly as a result, highly severe fire effects often concentrate at upper slope positions and on ridges (Taylor and Skinner 1998).

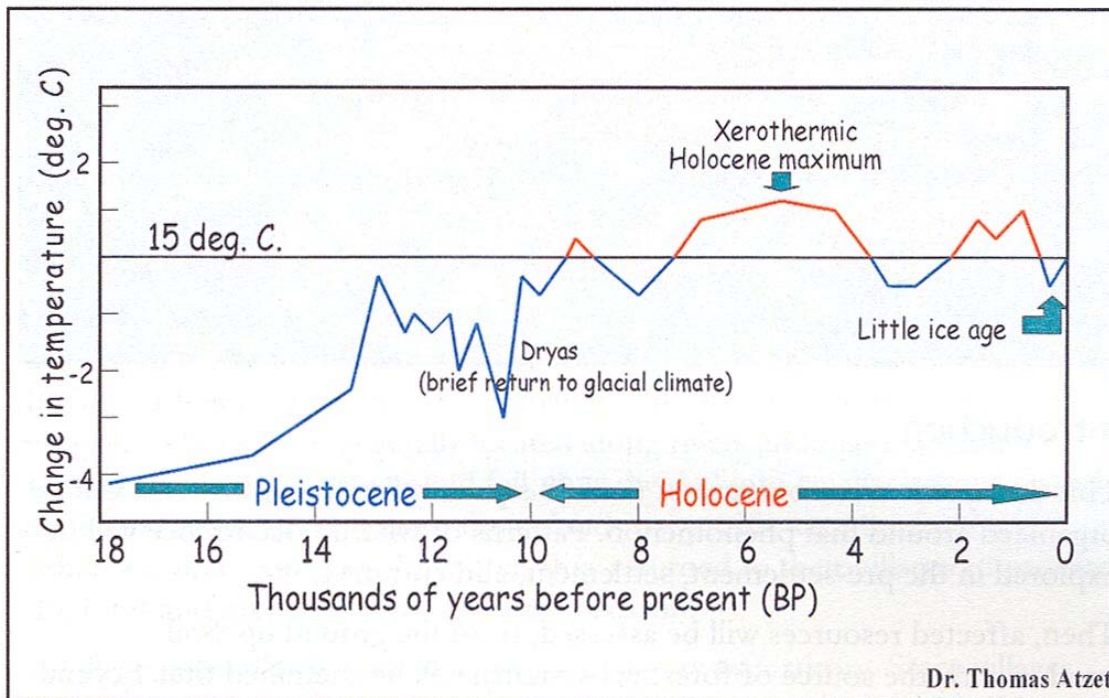
Variation in the duration of time elapsed between fires (return interval or rotation) reflects spatially heterogeneous landscape patterns and is scale-dependent (Taylor and Skinner 2003). For example, Willis and Stuart (1994) report point sample fire return intervals ranging from three-to-71 years over a 350-year period at three sites totaling 20 hectares on Hotelling Ridge near Forks of Salmon, California. Taylor and Skinner (1998) describe point sample return intervals ranging from five-to-116 years over a similar period at 75 sites totaling 1,570 hectares along Thompson Ridge near the California-Oregon border. Longer fire return intervals generally occur at mesic locations such as

northerly slopes and riparian areas in valley bottoms, whereas fires tend to burn more frequently on hotter, drier and more exposed southerly aspects, especially on upper slopes and ridges exposed to lightning (Agee 1993).

The result of spatial and temporal variation in the fire regime of mixed evergreen forests is a patchy landscape of stands exhibiting different tree densities, ages and species mixtures that continuously change over space and time (Willis and Stuart 1994). Forest stands exhibiting structurally diverse late-successional conditions develop with recurrent fires of low and moderate severity, and occasionally experience severe stand-replacing fires (Taylor and Skinner 1998). The latter initiate early-successional communities rich with coarse wood that enhance the age-structure mosaic of the landscape (Lindenmayer and Franklin 2002, Turner et al. 1998, White 1979). Highly severe fire effects account for significant proportions of the total area burned in virtually all recorded fire events in the KS region (Frost and Sweeney 2000), and they make significant positive contributions to landscape pattern and biological diversity (Martin and Sapsis 1992, Smucker et al. 2005).

### *Climatic variation*

Climatic influences compound temporal variation in the landscape fire regime of mixed evergreen forests. The global climate is dynamic (Figure III-2) and planetary scale temperature shifts affected the historical environment of southwest Oregon over millennia. Shortened fire frequency coincides over decadal timescales with periods of regional drought that encourage ignitions (Agee 1993). Over centennial and millennial timescales, climatic oscillations driven by shifts in solar radiation, orbital proximity of Earth to the sun, and the spatial distribution of polar ice caps influence continuous



**Figure III-2.** Global change in temperature from 18,000 years BP to the present time. Note the coincidence of the “Little ice age” with the recruitment period (~300 years BP) of existing mature forests in the Siskiyou Mountains. Source: USDA (2004).

changes in fire frequency at regional and local scales, confounding its predictability (Alaback et al. 2003).

Paleoecological research conducted in the Pacific Northwest suggests a climatic sequence relevant to the study area (Thompson et al. 1993, Whitlock 1992). However, no direct evidence informs characterization of climatic influences on vegetation, fauna, hydrology or human use patterns in southwest Oregon. Application of the following general outline of climate history to the study area must be treated cautiously because fine-scale spatial variations mediated by local terrain can affect temperature and precipitation patterns (Mock and Bartlein 1995).

Following deglaciation ~12,000 years BP the northern hemisphere experienced an amplified seasonal cycle of solar radiation lasting until ~6,000 BP. During this period solar radiation was greater in summer and winter radiation was less than today, resulting in warmer temperatures and less effective annual moisture. This xeric period eventually moderated and cooler, wetter conditions predominated after ~4,000 years BP. The timing of this transition varied throughout western North America, and direct evidence for southwest Oregon is lacking.

Shifting temperature and precipitation patterns throughout the Holocene may have affected the type and extent of dominant vegetation, stream flows, animal populations and human activities in southwest Oregon (Winthrop 1993). Contemporary patterns of climate and vegetation emerged over the past two millennia, although drought and wet periods continually vary in duration and intensity. Tree ring data sampled from forest stands in the upper Applegate River watershed hint that a severe, prolonged drought occurred in the middle third of the 18<sup>th</sup> century and repetitive dry climate cycles lasting five-to-10 years continued throughout the 19<sup>th</sup> century (LaLande 1995).

Fire disturbance and forest succession processes function in disequilibrium with climate (Allen et al. 2002). Therefore, existing plant communities with older vegetation may reflect recruitment responses to climatic conditions that no longer exist (Millar and Woolfenden 1999). Mann and colleagues (1999) show that the last decade of the 20<sup>th</sup> century was the warmest of the past millennium. For this reason, Mock and Brunelle-Daines (1999) suggest that the relatively xeric climatic conditions of ~6,000 years BP may offer a better analogue to contemporary climate than mesic conditions that prevailed in the past two centuries.



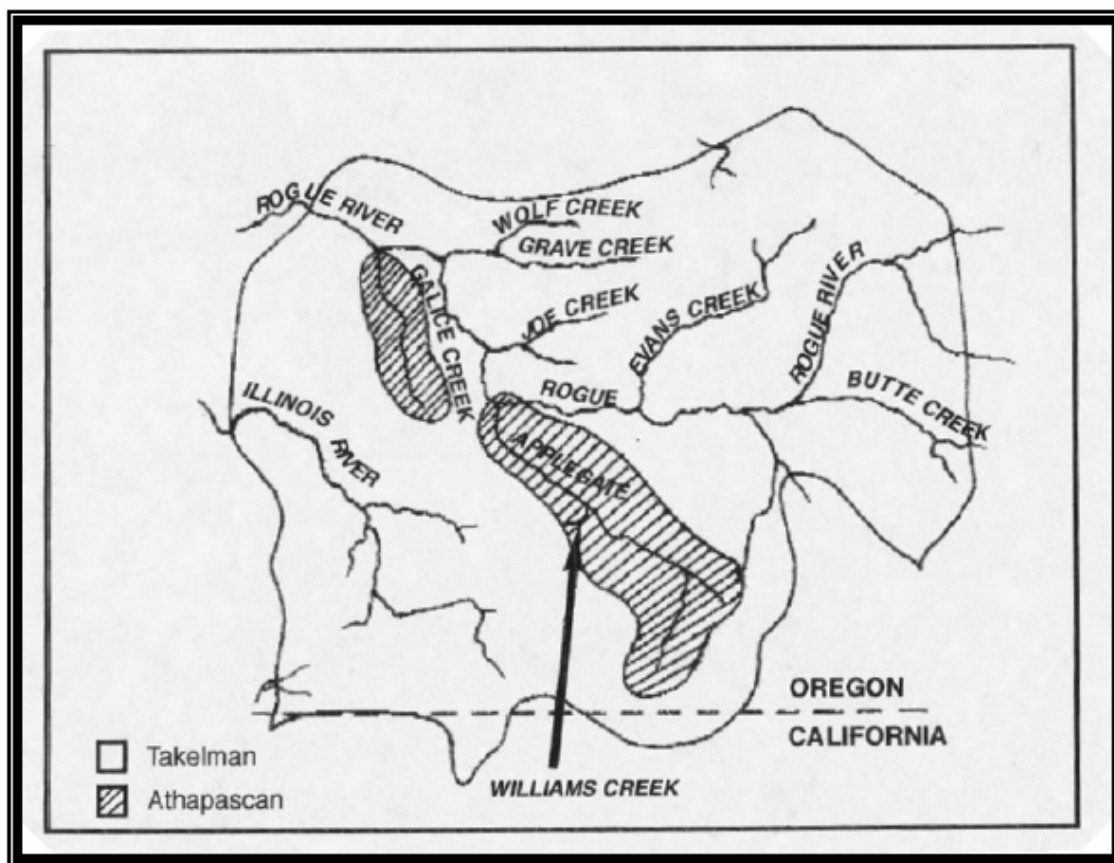
### *Indigenous human activities*

The “fire-climax” Whittaker (1960) observed in mixed evergreen forests of the Siskiyou Mountains reflected legacies of anthropogenic burning. The Applegate River watershed historically was home to Dakubatede people who actively managed their environment for subsistence (Beckham 1971, LaLande 1995, Pullen 1996, Winthrop 1993) (Figure III-3). Highly skilled and cyclical applications of fire technology kept selected locations open for hunting and gathering, stimulated berry and seed production, produced quality wildlife forage, reduced disease and insect infestations, and secured food supplies as well as fibers and medicinal plants (Kimmerer and Lake 2001).

Indigenous fire use also maintained ecotones among different vegetation communities, such as those linking savannah and forest, which supported high levels of biological diversity and were especially productive of foods and materials used by people (Boag 1992). Kimmerer and Lake (2001:38) argue that applied fire practices included

the intentional creation of a mosaic of habitat patches that promoted food security by ensuring a diverse and productive landscape [...] Maintaining a diversity of habitats buffers the impact of natural fluctuation in a single food species and increases overall productivity [...] Indigenous people skillfully modified the fire regime to create a range of forest openings in many different stages of postfire succession, which enhanced the diversity and yield of game, berries, root crops, edible seeds, and medicinal plants.

Indeed, food supply diversity was critical to survival of the Dakubatede because they routinely faced starvation in early spring before plant growth began and before the spring salmon run (Baumhoff 1963). The sap and inner bark of sugar pine and ponderosa pine were particularly important food resources during times when other foods were not available. California black oak acorns also were favored in good crop years due to their



**Figure III-3.** Geographic range of indigenous Dakubatede people in southwest Oregon. These people spoke a coastal Athapascan dialect, which distinguished them culturally from neighboring Takelma and other human communities. Source: Beckham (1971).

high fat content and ease of harvest in the foothills and lower edges of conifer forests, especially on south-facing slopes (Pullen 1996).

Use of fire by indigenous people helps to explain the historical dominance of sugar pine in certain low-elevation woodlands, which Kimmerer and Lake (2001) suggest could not be sustained solely by a lightning-ignited fire regime. Ethnobotanical practices of the Karok people in the nearby Klamath River canyon traditionally employ regular applications of fire to reduce competition and promote vigor in sugar pine stands (Schenk and Gifford 1952). The Karok seasonally shared the Siskiyou high country and intermarried with their Dakubatede neighbors, who led similar lifestyles (Pullen 1996)

(Figure III-3). Therefore, it is reasonable to propose that the Dakubetede employed similar fire use practices as the Karok to promote sugar pine at foothill locations in the Applegate River watershed.

Plot data indicate that very large (>60 cm dbh) sugar pines once were numerous and widespread (~20 stems/ha) throughout Unit 1 of the study area. Logging operations in the mid-20<sup>th</sup> century removed most of these specimens and left stumps that today are identifiable by their remnant bark. The unit is bound to the north and south by perennial creeks that would serve as effective containment lines for human-ignited prescribed fires. The topographic location of Unit 1 combined with its proximity to known historical settlements and its high concentration of large sugar pine and other nutritious plants (Table III-1) support the hypothesis that it may have been a fire-cultivated food production site.

Woodland tarweed (*Madia madioides*) (Figure III-4) is another important fire-cultivated food source observed throughout the study area. European settler George Riddle (1953:46) recalled tarweed seed cultivation in the south Umpqua River watershed 40 km north of the study area:

During the summer months the squaws would gather various kinds of seeds of which the tarweed seed was the most prized. The tarweed was a plant about thirty inches high and was very abundant on the bench lands of the valley, and was a great nuisance at maturity. It would be covered with globules of clear



**Figure III-4.** Tarweed (*Madia madioides*) was an important fire-cultivated food source for the Dakubetede people of the Applegate River. Source: Abrams and Farris (1960).

tarry substance that would coat the head and legs of stock as if they had been coated with tar. When the seeds were ripe the country was burned off. This left the plant standing with the tar burned off and the seeds left in the pods. Immediately after the fire there would be an army of squaws armed with an implement made of twigs shaped like a tennis racket. With their basket swung in front, they would beat the seeds from the pods into the basket. This seed gathering would only last a few days and every squaw in the tribe seemed to be doing her level best to make all the noise she could, beating her racket against the top of her basket.

People would grind harvested tarweed seeds into flour and cook them into a rich soup (Pullen 1996). Larger seed fragments were eaten or used to flavor manzanita cider; the roots also were eaten (Stevens and O'Brien 2001).

Climatic change from a hot and dry regime favoring oak-pine woodlands and grasslands to a milder and wetter regime more hospitable to fir forests ~4,000 BP may have forced technological adaptations among indigenous people who depended on natural resources common to the prior climate (Winthrop 1993). For this reason, archaeologists and ethnographers argue that cyclical burning in portions of the landscape originated as an attempt to manipulate or prevent an advance of fir forests into oak-pine woodlands and savannas that sustained the food supply (ARWC 1994, Pullen 1996, Winthrop 1993).

#### *Settlement era condition*

Concurrent with the genocide of indigenous people that accompanied European settlement in southwest Oregon (Beckham 1971), surveyors contracted by the U.S. General Land Office (GLO) subdivided and mapped the territory in a rectangular grid. They recorded “the several kinds of timber and undergrowth, in the order in which they predominate,” in hand-written field notes and maps (White 1983). Notes from 1856-57 surveys in T. 38 S., R. 5 W., Willamette Meridian, describe forest vegetation at the station dividing sections 3, 4, 9 and 10, approximately one kilometer (km) west of the

study area, as “pine, oak, fir & laurel.” A map appended to those notes establishes that the general distribution of this vegetation included the study area (GLO 1857).

A later GLO (1916) survey produced finer-scale information about forest structure and composition in the study area. It describes the western (upslope) half of Unit 1 as “covered with scattered timber and patches of brush.” Large specimens of sugar pine and ponderosa pine 60-75 cm in diameter occupied the upper canopy. In contrast, the eastern (downslope) half of Unit 1 was “generally free from brush” and hosted “few scattered trees,” mostly pine.

The same record also describes “light stands of timber and brush” on the north face of Pennington Mountain, where Unit 2 is located (GLO 1916). Douglas-fir was two-to-six times more abundant in the western half of Unit 2 than sugar pine or ponderosa pine. However, pines dominated the forest composition on relatively flat and west-facing slopes in the unit’s eastern portion. Pines also were more abundant in the southern (upslope) portion of the unit than in its northern (downslope) portion. Trees of all conifer species exhibited >60 cm diameter stems.

#### *Post-settlement changes*

The displacement of indigenous people and concurrent introduction of a European industrial economy in the mid-19<sup>th</sup> century transformed landscape patterns in the Siskiyou Mountains. The changes included construction of an extensive road network, replacement of forests dominated by large, widely-spaced trees with dense stands of tightly-spaced poles and saplings, and significantly less area burned by fire (USDA/USDI 1994a). Resulting fragmentation and compositional changes in plant communities

reduced landscape heterogeneity and biological diversity, particularly in low elevation valleys and foothills (Staus et al. 2003).

Forest composition and structure in the study area reflect the above described changes in human use and disturbance patterns. Roads fragment the area (Figure II-3), and high-grade logging in the mid-20<sup>th</sup> century removed many of the largest trees, especially pines, leaving densely packed stands of smaller trees (USDI 1996). Cumulative effects of logging and fire suppression promoted growth of less fire-tolerant species such as Pacific madrone and white fir (USDA/USDI 1994a). Continuous vegetation structure is now common and tree vigor is in decline as a result of overstocking (USDI 1999). Dense stands and accumulated surface fuel, particularly duff, increase the likelihood of severe fire effects relative to historical conditions (Arno and Fiedler 2005). Moreover, a proliferation of residential dwellings creating a rural-wildland interface adjacent to the study area compounds this general decline in ecological integrity. Elevated risk of ignitions causing catastrophic loss of values at-risk presents a significant challenge to ecosystem management (USDI 1996, 2000).

#### *Means to achieve reference conditions*

Knowledge of historical conditions in the study area can facilitate restoration efforts, but attempts to recreate conditions that existed in the recent past would not be desirable or feasible for many reasons. First, the climate is different than it was before, and it will continue to change (Millar and Woolfenden 1999, Whitlock et al. 2003). Second, historical documentation offers useful perspective on large tree structure and composition, but it is not as reliable for small trees, herbaceous vegetation or wildlife use

(Harrod et al. 1999). Finally, fire regimes of the past can be estimated but are poorly understood, and these, too, shift over time (Agee 1998, Baker et al. 2006, Veblen 2003).

Abundant evidence suggests that wildland fire will become more frequent and severe at a landscape scale as the regional climate becomes more xeric (McKenzie et al. 2004, Pierce et al. 2004, Running 2006). Therefore, a sensible approach to stand-scale forest restoration in the study area would prepare the site for inevitable fire disturbance and attempt to facilitate, insofar as possible, self-sustaining vegetation density, structure and composition analogous to conditions in which ecosystem functions evolved (Arno and Fiedler 2005, SER 2004). Historically dominant tree species such as sugar pine, ponderosa pine and California black oak play important roles in the ecological function of forest and woodland communities in the study area (USDI 1996). Those species require open stand conditions and frequent disturbance to flourish (Atzet et al. 1996).

In the long-term, reintroduction of fire is the most important action to restore ecological integrity because it is the only way to stimulate adapted interactions between ecosystem structure and process (McIver and Starr 2001). Fire disturbance plays a vital role in supporting a diverse complement of plant species and structure, and maintaining these qualities over time is essential to support terrestrial and riparian processes impacted by industrial uses and fire suppression (Atzet 1996). Fire restoration will be most effectively accomplished at landscape scales (Baker 1994), but localized restoration treatments may help to influence the intensity of fire behavior and severity of fire effects. This is especially important in the study area where rural residences and other human developments interface with fire-prone wildland vegetation (USDI 1996, 2000). In this interface, fuel reduction treatments should be coordinated with efforts to “fire-proof”

homes by addressing their ignitability (Cohen 2004). These initial steps are necessary before land managers realistically can undertake ecologically-based fire restoration treatments at broader spatial scales (Brown et al. 2004).

Thinning of small, shade-tolerant and suppressed understory trees coupled with management-ignited prescribed fire (MIPF) has rendered some woodlands and forests less susceptible to uncharacteristically severe fire effects (stand replacement) (Perry et al. 2004, Pollett and Omi 2002). Primary variables accounting for canopy fire initiation include surface fuel load, live fuel moisture and canopy base height (Agee et al. 2000). Therefore, thinning of small trees “from below” to increase canopy base height followed by reduction of activity fuels (slash) and pre-existing surface fuels can reduce canopy ignition potential (torching) (Graham et al. 2004).

Fire managers in the KS region often prioritize reduction of canopy bulk density to limit potential for active canopy fire spread (e.g., USDA 2004, USDI 2005). Canopy fire behavior depends on the density of canopy fuels as well as the availability of surface fuels to sustain convective heat transfer into the canopy (Scott and Reinhardt 2001). In order for active crown fire to spread independent of surface fireline intensity, it needs rare combinations of topography and weather, and even then, it usually affects only small areas before ceasing on its own (Van Wagner 1977). In Unit 1, flat terrain mitigates the hazard of active canopy fire. Effective fuel treatments to improve stand resilience there will reduce surface fuels and increase canopy base height to minimize surface fire intensity and diminish convective heat transfer into canopy fuels. Steeper terrain in the upslope portion of Unit 2 increases the chance of active canopy fire in extreme weather conditions, and canopy bulk density reduction there could help to alleviate it (Graham et



al. 2004). However, as discussed below, that portion of Unit 2 where slopes exceed 30% host relatively mesic conditions as well as rare and sensitive clustered lady's slipper orchid (*Cypripedium fasciculatum*), which associates with closed canopy forest habitat on northerly slopes in the Applegate River watershed (USDA/USDI 1994b). Therefore, the BLM is not likely to undertake significant reductions of canopy bulk density at that location. Further, investigations of historical fire severity patterns at stand and landscape scales in the KS region correlate low severity fire effects with closed canopy forest structure and high severity fire effects with structural conditions created by crown bulk density reduction treatments (e.g., Odion et al. 2004, Raymond and Peterson 2005).

Mechanical tree cutting in any form cannot replicate many ecological functions of fire, and approaches that rely solely on manipulating forest structure without restoring fire process will not achieve ecologically beneficial outcomes (DellaSala et al. 2004). Nevertheless, thinning can be an appropriate initial step toward fire restoration where existing forest structure precludes reintroduction of fire, and where the risk of adverse ecological impacts posed by thinning is relatively low (Franklin and Agee 2003). Such places generally occur on dry sites at low elevations with flat terrain and stable soils (Brown et al. 2004). Roads and evidence of past logging are obvious, and forests are dense relative to historical conditions, as shown by site-specific evidence indicating the prior existence of a low-severity fire regime (DellaSala and Frost 2001).

Unit 1 of the study area exhibits the above specified qualities that justify mechanical thinning to manipulate forest structure and composition as an initial step toward longer-term fire process restoration. A credible restoration-oriented thinning prescription and subsequent fuel treatments in Unit 1 would:

- Thin small, shade-tolerant, and suppressed trees (generally <30 cm dbh) of species that are relatively abundant as a result of logging and fire exclusion. Thinning should strive to reduce short-term hazard of stand replacing fire and create conditions in which fire can safely be used over longer periods of time to support, insofar as possible, adapted ecological processes (USDI 2002).
- Protect large, old trees from removal or damage. These legacy structures possess fire resilient qualities, serve numerous important ecological functions, are difficult or impossible to replace, and are less abundant now than they were in the past (Brown et al. 2004, Franklin and Agee 2003).
- Create relatively open stand structure below the upper canopy because this condition would be expected under an active fire disturbance regime at the site (USDI 1996).
- Reduce fine woody surface fuels (<7.6 cm) and accumulated duff, the latter of which can sustain extended fire residence times causing more severe fire effects on vegetation and soil compared to reference conditions (Harrington 2000). Fine woody fuels are likely to increase substantially after thinning operations as tree stems are severed and relocated to the ground. Therefore, post-thinning slash treatment is critical to achieve management goals (Table IV-1).
- Monitor the site immediately after restoration activities and periodically thereafter to determine treatment effectiveness and any need for additional follow-up treatments.

Large (>60 cm dbh) trees generally should not be removed from the study area, in part, because a distinguishing feature of ecologically healthy and resilient mixed evergreen forests in the KS region is the prevalence of large, old trees that have survived numerous fires (Willis and Stuart 1994). Such trees represent the best potential for recruitment of structural characteristics that historically existed on the site. This is not to suggest that an arbitrary upper diameter limit should be imposed on tree cutting, but merely to illustrate the importance of large tree retention.

Much of the preceding discussion also applies to Unit 2 of the study area, but its unique site characteristics merit qualification. The unit mostly occurs on a north-facing slope that historically was dominated by mature Douglas-fir except for an isolated pocket

of oak savannah on ultramafic soil near the unit center and a pine-dominated flat near its eastern boundary. Passive restoration, defined as cessation of practices that degrade ecological function (fire suppression), may be an appropriate restoration approach there (DellaSala et al. 2004). However, limited mechanical thinning and MIPF applications could be justified in the oak savannah, pine flat and at lower slope locations occupied by Douglas-fir with relatively flat terrain and stable soils. Perceived benefits of such operations, including training of inexperienced crews, must be weighed against the cost of active management in places that may self-restore if left alone. Thinning and burning would not be as appropriate on slopes >30% where botanical indicators of mesic conditions such as rare and sensitive clustered lady's slipper orchid exist.

#### *Effectiveness criteria*

Stand density, structure and species composition all indicate how forest and woodland communities respond to environmental stress (USDI 1999). In Unit 1, seral conifers and hardwoods with tall canopies should dominate, and drought-tolerant shrubs should be present on the ground surface. Specifically, the following species would be expected to occur in an approximately descending order of relative abundance:

- Sugar pine (*Pinus lambertiana*)
- California black oak (*Quercus kelloggii*)
- Ponderosa pine (*P. ponderosa*)
- Whiteleaf manzanita (*Arctostaphylos viscida*)
- Deer brush (*Ceanothus integerrimus*)
- Douglas-fir (*Pseudotsuga menziesii* var. *douglasii*)
- Pacific madrone (*Arbutus menziesii*)
- Poison oak (*Toxicodendron diversilobum*)
- California fescue (*Festuca californica*)
- Blue wildrye (*Elymus glaucus*)

Stand density in Unit 1 should be more open than in the pre-treatment condition with an initial maximum relative density index of 0.5, and trending to lesser density over time to promote pine and oak regeneration (USDI 1999). Vertical stand structure ought to feature a mean canopy base height >3 m tall and minimal connectivity with surface vegetation. Dominant and co-dominant canopy layers should include gaps over a significant proportion (>25%) of the stand to promote sunlight penetration to the ground surface and growth of forbs and grasses.

The amount, continuity and moisture content of fine surface fuels determine the rate at which wildland fire spreads and the intensity with which it releases heat energy (Rothermel 1983). Reference condition surface fuel loading is <5 kg/m<sup>2</sup> throughout Unit 1 and, as above, canopy base height is >3 m tall resembling a Fuel Model 8 (Anderson 1982). In this condition surface fuel moisture, ambient temperature and horizontal wind movement can support intensified flaming heat energy release potential and faster rates of fire spread compared to the pre-treatment condition, but stand replacement is less likely due to structural limitations on canopy fire initiation.

In Unit 2, seral conifers and hardwoods with tall canopies also should dominate the stand, although understory plants should be more representative of mesic conditions on northerly slopes at slightly higher elevations than Unit 1. The following species would be expected to occur in an approximately descending order of relative abundance:

- Douglas-fir (*Pseudotsuga menziesii* var. *douglasii*)
- Poison oak (*Toxicodendron diversilobum*)
- Canyon live oak (*Quercus chrysolepis*)
- Creambrush oceanspray (*Holodiscus discolor*)
- Hairy honeysuckle (*Lonicera hispidula*)
- Creeping snowberry (*Symphoricarpus albus*)
- Ponderosa pine (*Pinus ponderosa*)
- California black oak (*Q. kelloggii*)

- Sugar pine (*P. lambertiana*)
- California fescue (*Festuca californica*)

Stand density in Unit 2 should be more open than in the pre-treatment condition with a maximum relative density index of 0.6, trending to lesser density over time (USDI 1999).

Vertical stand structure ought to feature a mean canopy base height >2 m tall and minimal connectivity with surface vegetation. Dominant and co-dominant canopy layers should include gaps over a portion (>20%) of the unit, particularly in the central oak savannah and easterly pine flat, which should be more open (RD <0.4) than the steeper upslope and westerly portions dominated by Douglas-fir.

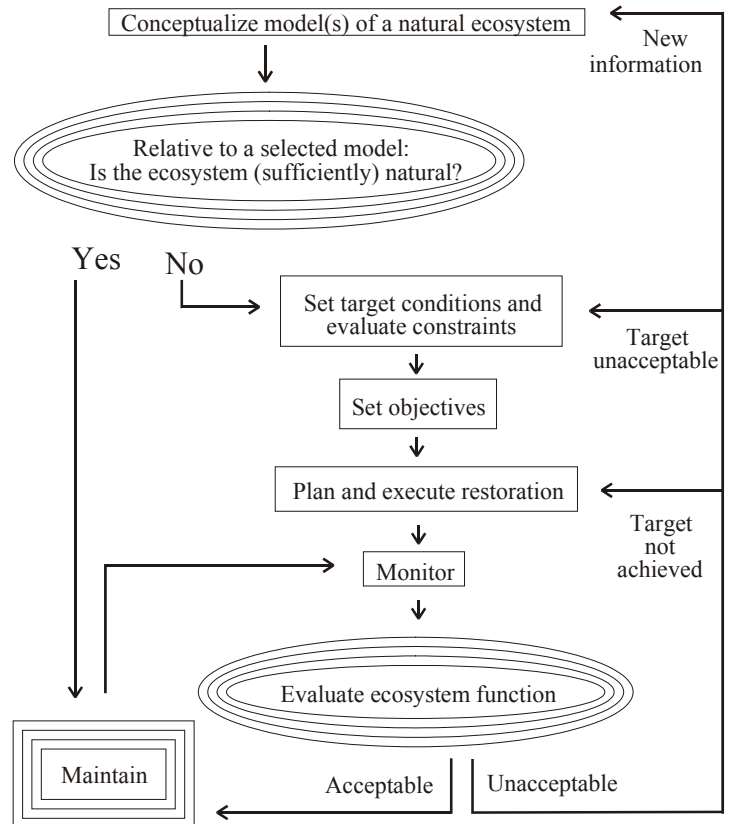
Overall, reference conditions in the study area encompass a range of potential conditions in which land managers would be more comfortable using wildland fire to achieve restoration of adapted ecological processes (Franklin and Agee 2003). For reasons outlined above, it is not realistic or desirable to expect a single treatment or prescription to realize the reference condition (Brown et al. 2004). Instead, a series of light thinning treatments coupled with applications of MIPF to reduce surface fuel loads, fuel ladders and live tree density can support longer-term restoration. Monitoring is essential to verify effectiveness and inform assessment of follow-up treatments over time.

## IV. MONITORING REPORT

### Introduction

Monitoring is a centrally important process in adaptive ecosystem management. Adaptive management embraces scientific uncertainty and learning by doing (Lee 1993). The key to learning is a well-defined monitoring strategy that provides a “feedback loop” through which management effects and means to improve practices can be described (Figure IV-1). Paradoxically, mistakes and failure often contribute the most to success in terms of learning from the effort. This emphasis on learning means that failures do not exist in the traditional sense (Lee 1993).

The Northwest Forest Plan (USDA/USDI 1994b) established an Adaptive Management Area (AMA) in the Applegate River watershed of southwest Oregon with an objective of testing non-traditional forest practices that integrate ecological, economic, and social objectives. The inherently controversial nature of wildland management in the AMA (USDI 1999) creates an incentive for forestry practitioners to clearly



**Figure IV-1.** Model of the adaptive feedback process in ecological restoration. Source: Keeley and Stephenson (2000).

document the results of their actions for public understanding. Indeed, the purpose of the AMA is to “provide a geographic focus for innovation and experimentation with the intent that such experience will be widely shared” (USDA/USDI 1994b: D-2).

Wildland managers and forestry practitioners often forego monitoring that supports adaptive management. Some find themselves too busy with other essential duties to design and implement monitoring projects. Moreover, the perceived complexity of sampling designs necessary to obtain useful monitoring data can overwhelm or intimidate some workers. Perhaps the most important reason why credible monitoring projects rarely get implemented is a lack of standardized sampling methods and tools. Most organizations have not developed science-based sampling protocols to inventory pre- and post-activity conditions and document achievement of management goals. The major exception is the National Park Service (NPS), which has developed extensive guidelines for sampling ecosystem characteristics important to monitoring (USDI 2003). Others increasingly use the FIREMON fire effects monitoring system (Lutes 2006), which is designed to be flexible to users’ differing needs for rigor and repeatability.

This report describes an effectiveness monitoring strategy for the “Penny Stew” forest stewardship project undertaken by the Lomakatsi Restoration Project (LRP) on federal land in the Applegate AMA controlled by the U.S. Bureau of Land Management (BLM). It defines management goals and monitoring objectives, the methods used to monitor activity effects and evaluate data, and it provides feedback with initial treatment results and discussion. Specifically, it compares observed treatment effects with stated management goals, and it evaluates methodological efficacy for consideration in later monitoring efforts. The intent is to demonstrate a credible and repeatable strategy that the

LRP can adapt to learn by doing, refine its forest practices, and demonstrate its activity effects to interested parties.

Furthermore, this analysis tests the efficacy of hybridized monitoring protocols that combine the permanent monitoring plot scheme of the NPS system (USDI 2003) with sampling methods and data analysis tools of the FIREMON system (Lutes 2006). This approach blends the most useful elements of each system into a streamlined protocol that can be implemented by practitioners with little formal education or training. Since complexity often is a decisive barrier to sustained monitoring, it is important to find ways to simplify the task without compromising defensibility.

Resource availability imposes additional limitations on monitoring. Practitioners' curiosity frequently exceeds their ability to fund the requisite sampling and analysis to address all conceivable avenues of inquiry. This analysis presents the minimum information needed to quantitatively address core management goals related to forest structure, composition, and fire hazard. Furthermore, it offers visual evidence of pre- and post-treatment conditions represented by permanent photo points to facilitate qualitative evaluation of changing conditions over time.

The Penny Stew contract divides the area of federal land included in the project ("study area") into two units (Chapter II). This report documents the monitoring effort in Unit 1, where planned treatment activities were completed by the time of this writing. It does not address monitoring in Unit 2 because no results other than pre-treatment baseline data were available. This report should be viewed as iterative, as it is intended to be supplemented and updated as treatment activities and monitoring progress.



Management goals for the Penny Stew project (Table IV-1) are based on the *Scattered Apples Forest Management Project Environmental Assessment* (USDI 1999) and *Decision Record* (USDI 2002), which govern activities in the study area. Those documents mandate thinning in the forest understory to reduce tree density and decrease competition for nutrients, water, and light among residual trees. Generally, only trees smaller than 30 centimeters (cm) in diameter at breast height (dbh) may be severed or removed. Trees larger than that size are to be retained except where retention may pose a safety hazard to field workers. Tree species favored for retention are described in the accompanying reference analysis (Chapter III).

Fire hazard reduction goals also drive the Penny Stew project. Hand piling and burning of pre-existing surface fuels and thinning slash are to remove 50 to 75% of downed woody fuels 3 to 15 cm in diameter over a period of 10 years (USDI 2002). Fuel particles outside this size range are to be left scattered so that they contact the ground surface. The treatment prescription should leave a more compact fuel bed that would slow rates of fire spread and shorten flame lengths. It also should decrease the time required for decomposition of downed woody debris remaining on the ground (USDI 2002).

In addition to the goals described above, the LRP specified complementary goals. First, it seeks maintain effective soil cover and minimize erosion as a result of treatments. Second, the LRP desires to detect and discourage exotic species with potential to degrade ecological function such as cheat grass (*Bromus tectorum*) and yellow star thistle (*Centaurea solstitialis*). Finally, it strives to nudge forest density, structure, and composition to better resemble reference conditions (Chapter III) while embracing a

precautionary preference for conservative and incremental treatments that “do no harm” rather than pursuing more aggressive treatments that would impose reference conditions with a single treatment (Brown et al. 2004). It is thus expected that follow-up treatments and periodic maintenance will be required to attain management goals in the study area. This report does not address future treatments, which will require additional planning and assessment. However, it presents repeated measurements of selected indicators in permanent plots to inform consideration of maintenance or other follow-up treatments as needed (Figure IV-1).

## **Method**

### *Monitoring objectives*

We identified site-specific management goals expressing target stand conditions (Table IV-1) based on the stated purpose and need for action (USDI 1999) and the results of reference analysis (Chapter III). We then defined monitoring objectives that identify measurable indicators of each management goal as well as the minimum detectable change (MDC), confidence level, and timeframe relevant to sampling design power analysis and evaluation of results (Table IV-2).

Woodward and others (1999) suggest criteria for monitoring indicator selection emphasizing logical linkages to management goals, ease of field identification, sampling cost-effectiveness, importance to ecosystem function, and social appeal. Table IV-1 shows the selected indicators relevant to management goals for the Penny Stew project. Those indicators meet the above criteria and fit the purpose of this analysis to demonstrate a streamlined protocol that is easy and efficient to implement.

Monitoring objectives (Table IV-2) specify a minimum detectable change (MDC) and a confidence level (generally 80%) for change detection in each indicator. The MDC and confidence level function as guidelines for sampling design power analysis (Figure IV-2) and evaluation of results. The iterative nature of monitoring objectives is a tenet of adaptive management (Elzinga et al. 1998). Monitoring objectives may change over time as new information emerges or if management goals shift.

**Table IV-1.** Management goals, indicators and target conditions.

Management goal	Indicator	Target condition
Promote reference composition	Species abundance	Pine/oak stems >30% of all trees in 10 years Douglas fir stems <40% of all trees in 10 years
	Legacy vegetation	Large (>50 cm dbh) trees and snags maintained over 10 years
Promote reference structure	Tree density	Relative density index <0.5 in 5 years, <0.4 in 10 years
	Mature tree diameter	Mean live tree diameter >30 cm dbh in 10 years
	Mature tree basal area	Basal area <20 m <sup>2</sup> /ha in 10 years
Promote fire resilience	Surface fuel load	Surface fuel load <6 kg/m <sup>2</sup> in 5 years, <4 kg/m <sup>2</sup> in 10 years
	Live fuel density	Sapling density <500 trees per hectare in 5 to 10 years
	Mature tree diameter	Mean live tree diameter >30 cm dbh in 10 years
	Canopy base height	Canopy base height >3 m tall in 5 to 10 years
Maintain effective soil cover	Cover percent	>80% average soil cover maintained over 10 years
	Cover depth	>3 cm average cover depth maintained over 10 years

**Table IV-2.** Management indicators and monitoring objectives.

Management indicator	Monitoring objective
Species abundance	Detect with 80% confidence a 30% reduction of Douglas fir stems in 2 to 5 years
Tree density	Detect with 80% confidence a 30% reduction of total stem density in 2 to 5 years, and 50% in 10 years
Mature tree diameter	Detect with 80% confidence a 20% increase of mean live tree diameter in 2 to 5 years
Mature tree basal area	Detect with 80% confidence a 20% reduction of mature tree basal area in 2 to 5 years
Surface fuel load	Detect with 80% confidence a 20% change of surface fuels in 2 years, and 50% in 5 to 10 years
Live fuel density	Detect with 80% confidence a 50% reduction of sapling density in 2 years, and 80% in 5 to 10 years
Canopy base height	Detect with 80% confidence a 20% increase of crown base height in 2 years, and 30% in 5 to 10 years
Cover percent	Detect with 80% confidence a 20% reduction of soil cover in 2, 5 and 10 years
Cover depth	Detect with 80% confidence a 20% reduction of cover depth in 2, 5 and 10 years

### *Pilot sampling*

Pilot sampling helped to define appropriate sample size, which in turn refined monitoring objectives and power estimates. We established three 0.1 ha (1000 m<sup>2</sup>) plots (Figure IV-3) in Unit 1 and collected data on forest structure and surface fuel variables in the manner described below. We then calculated means and standard deviations of pilot sample attributes, and estimated the sample size needed to fulfill monitoring objectives with the desired power ( $\alpha = 0.2$ ,  $\beta = 0.2$ ) by entering those values into the following equation supplied by Lutes (2006):

$$NRP = 2 \frac{s^2 (Z_{(\alpha f_2)} + Z_{(\beta f_2)})^2}{MDC^2}$$

Where:

- *NRP* is the number of required plots.
- *s* is the estimated standard deviation of mean sample values.
- $Z_\alpha$  is the coefficient for the type-I error rate in Table IV-3 below.
- $Z_\beta$  is the coefficient for the type-II error rate in Table IV-3 below.
- MDC is the minimum detectable change in mean sample values among sampling events sought by the monitoring objectives.

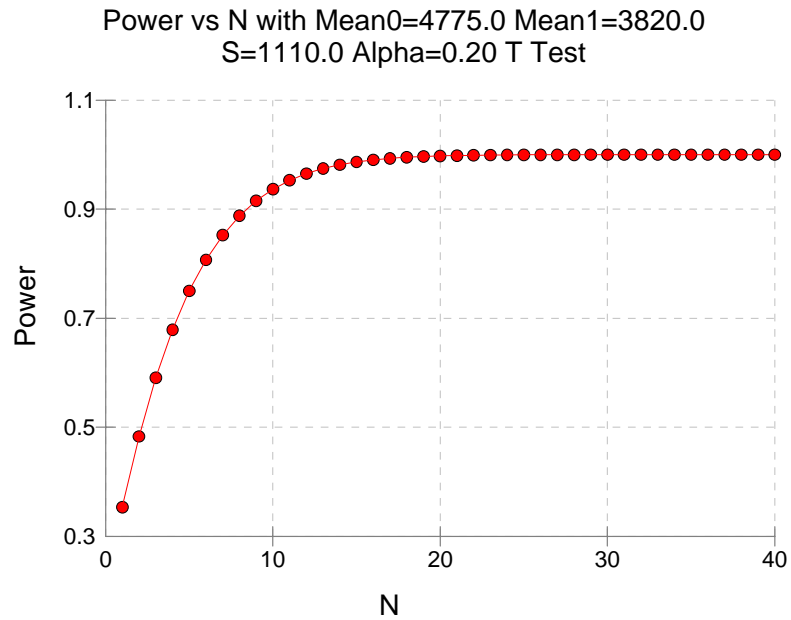
**Table IV-3.** Coefficients of acceptable error rates used to determine sample size.

False-change (type-I) error rate ( $\alpha$ )	$Z_\alpha$	Missed-change (type-II) error rate ( $\beta$ )	$Z_\beta$
0.4	0.84	0.4	0.25
0.2	1.28	0.2	0.84
0.1	1.64	0.1	1.28
0.05	1.96	0.05	1.64
0.01	2.58	0.01	2.33

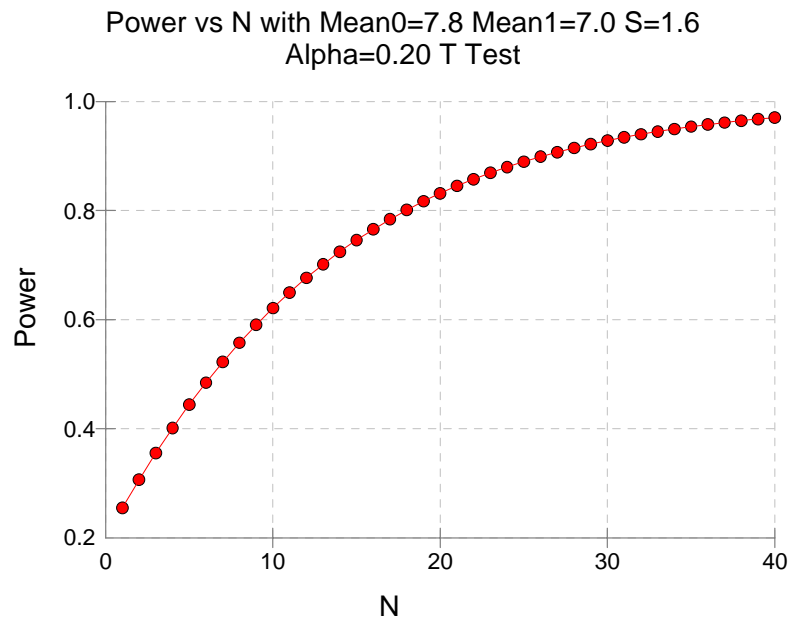
The equation shown above suggested that a relatively small sampling population ( $n=3$ ) of 0.1 ha plots would detect a 20 percent change of tree sapling density with 80 percent confidence, and a larger population ( $n=16$ ) of 20 meter (m) planar intercept transects (Brown 1974) would detect a 10 percent change of surface fuel load with similar power.

We cross-checked the sample size and power estimates above by entering pilot sampling values into NCSS/PASS software (Hintze 2004), which runs one-mean inequality tests with the same acceptable error rates, estimated standard deviations, and MDC parameters as above. The NCSS/PASS run indicated that doubling the sampling population ( $n=6$ ) would better fulfill the monitoring objectives for tree density and a similar population as indicated by the FIREMON equation above ( $n=18$ ) would suffice to detect change in surface fuels (Figure IV-2). Our calculations assumed that 0.1 ha plots would sample a finite population of 162 potential plot locations in the 16.2 ha unit, and that randomly directed planar intercept transects would sample an infinite population of surface fuels. Given the different sampling intensity estimates derived from the FIREMON equation and the NCSS/PASS computations, respectively, we implemented the greater intensity level suggested by the latter in an effort to generate more robust results.

(a) Tree density



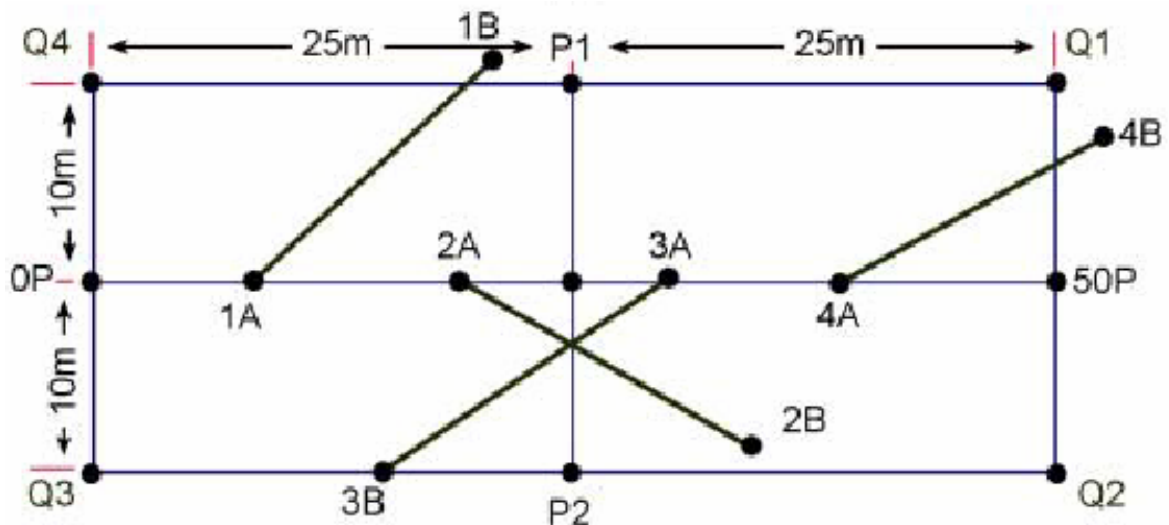
(b) Surface fuel



**Figure IV-2.** Results of NCSS/PASS power analysis computations used to determine sample intensity. Pilot sampling data suggested that (a) that six 0.1 ha plots would detect a 20% reduction of tree density with 80% confidence (power = 0.80714), and (b) 18 planar intercept transects could detect a 10% change of surface fuel load with similar power (power = 0.80133).

### Plot location and design

We established two parallel grid lines extending the length of Unit 1 (Map II-3) and marked origin points at fixed 150 m intervals along each line. We located plot anchor points at random distances and azimuths from each origin. We oriented the rectangular 0.1 ha plots (Figure IV-3) with their 50 m sides perpendicular and 20 m sides parallel to magnetic north. We then established two sampling units overlaid on each other: ecosystem monitoring (EM) plots from the NPS fire monitoring protocol (USDI 2003) and Modified-Whittaker (MW) nested subplots (Stohlgren et al. 1995). Rectangular EM plots are effective for measuring forest density and structural properties, and MW subplots are useful for sampling understory vegetation data because it is collected at multiple scales (Bonham 1989). Since the plots are intended to be permanent, this design is more efficient to lay-out, measure, relocate, and repeat than circular variable-radius plots presented in the FIREMON literature (Lutes 2006).



**Figure IV-3.** 0.1 ha (1000 m<sup>2</sup>) EM plot. Anchor point is at plot center. 20 m planar intercept transects (1A/1B, etc.) measure surface fuel load. Source: USDI (2003).



### *Sampling procedures*

Within each EM plot, we applied three FIREMON vegetation sampling methods: tree data (TD), fuel load (FL), and cover/frequency (CF) (Lutes 2006). In addition, we established permanent photo points according to FMH protocols (USDI 2003).

#### Tree data

In TD sampling we recorded the species, diameter at breast height (dbh), crown class, height, live crown percent, and crown base height of all live “mature” trees (>10 cm dbh) throughout each plot. We recorded the species, dbh, height, and decay class of all dead mature trees at the same scale. We also recorded the species, dbh, height, and live crown percent of “sapling” and “seedling” trees (<10 cm dbh, differentiated by height) within randomly selected 0.025 ha (250 m<sup>2</sup>) subplots, the locations of which were recorded on field data forms for repeat measurements. Live crown percent expresses the percent of each tree bole that supports live vegetation based on the distance from the ground to the top of the live foliage. Crown base height expresses the height from ground to live crown base (i.e., height of the lowest live foliage). It does not express the height of dead materials, even though these also can affect vertical flame movement. The inclusion of live foliage and exclusion of dead material intends to limit subjectivity in field data collection.

#### Fuel load

In FL sampling we placed four 20 m planar intercept transects after Brown (1974) at randomly chosen azimuths from points located at fixed 10 m intervals along the plot centerline (Figure IV-3). We counted fine wood particles in the 1-hour (<0.6 cm

diameter) and 10-hour (0.61 – 2.5 cm) fuel classes on a 2 m section of each transect, and we counted 100-hour (2.51 – 7.59 cm) fine fuels on a 4 m section. We counted 1000-hour (>7.6 cm) coarse fuels on a 15 m section. We also measured duff and litter depth at two points, as well as the percent and type of soil cover in five equally divided intervals (Table IV-4).

**Table IV-4.** Measurements on 20 m planar intercept transect.

Variable	Sampling location
<b>Downed wood</b>	
1-hour (<0.6 cm)	2 m to 4 m (2 m total)
10-hour (0.61 – 2.5 cm)	2 m to 4 m (2 m total)
100-hour (2.51 – 7.59 cm)	2 m to 6 m (4 m total)
1000-hour (>7.6 cm)	2 m to 17 m (15 m total)
<b>Duff &amp; litter depth</b>	2 m, 5 m
<b>Soil cover</b>	2 m, 4 m, 6 m, 8 m, 10 m

#### Cover/frequency

We used the CF method to qualitatively estimate vegetation cover in each plot. We sampled vegetation cover in five MW quadrats, each one-square-meter (1 m<sup>2</sup>) in size, placed at fixed 5 m intervals along a 50 m transect beginning at either the 5 m or 15 m point along a 20 m baseline of the EM plot. In some cases, the transect location was subjectively placed to avoid FL planar intercept transects where overlap may have resulted in fuel bed trampling or other sampling bias. Within each quadrat we estimated the percentage of ground covered by individuals of each plant species based on a visual estimate of cover class suggested by Lutes (2006) (Table IV-5). The range of cover values are divided into broadly defined classes that minimize bias due to observer

error. Variation in cover estimates due to observer perception was further minimized by using visual aids for consistency and by standardizing estimates among observers in practice runs before data collection. The midpoint value of each cover class support numerical computations, though such analysis assumes that actual cover values are normally distributed around

**Table IV-5.** Plant cover class codes.

Code	Cover values
0	Zero percent cover
0.5	>0-1 percent cover
3	>1-5 percent cover
10	>5-15 percent cover
20	>15-25 percent cover
30	>25-35 percent cover
40	>35-45 percent cover
50	>45-55 percent cover
60	>55-65 percent cover
70	>65-75 percent cover
80	>75-85 percent cover
90	>85-95 percent cover
98	>95 percent cover

the midpoints, which cannot be verified by the data. We also noted the presence of other species observed outside the 1 m<sup>2</sup> quadrats at 10 m<sup>2</sup> and 100 m<sup>2</sup> scales. This method intends to document change in species cover and composition over time, but not to quantify statistically significant changes, due to the subjective nature of cover estimations. This report does not present monitoring results of CF sampling, as there were insufficient data to analyze at the time of this writing. Future iterations of this report will include CF data analysis.

### Photo points

We established permanent photo points at the northeast and southwest corners (Q1 and Q3 – Figure IV-3) of each plot. At each photo point in every sampling event, we took digital photographs of the plot center. Unit and plot numbers, date, and sampling event were recorded in a log that matches site and time information with photo sequences.

### *Monitoring cycle*

All plots in Unit 1 are to be sampled according to the following schedule: prior to the initial treatment (P1), after thinning but before slash clean-up (R1), within one year after slash treatment (R2), and again five years (R3) and 10 years (R4) after thinning. Any additional treatment beyond post-thinning slash disposal triggers a new monitoring cycle that follows the same schedule beginning at R1. In addition, one control plot outside the unit is to be measured prior to initial treatment (P1) and 10 years after treatment (R4).

### *Data management and analysis*

We input the sample data to the TD, FL and CF database tables in the FIREMON software (JFiremon). We generated summary reports and performed a line-by-line comparison of database entries with field data forms to ensure data quality. We then applied the FIREMON Analysis Tool (FMAT) program to process TD and FL data. FMAT reports variable attributes for each plot and sampling event, mean values and standard deviations among samples, and percent change among events (Lutes 2006). We used its calculations of mature tree basal area (BA) and quadratic mean diameter (QMD) to determine relative stand density (RD) with the formula:  $RD = BA \cdot QMD^{-0.5}$  (Curtis 1982).

FMAT-produced values for surface fuel load are subject to the assumptions discussed below. JFiremon calculates woody fuel biomass volume according to the equations presented in Brown (1974). Non-slash, composite values are used for quadratic mean diameter, non-horizontal correction, and specific gravity of fine (<7.6 cm diameter)

woody debris. Coarse (>7.6 cm) woody debris input to the database as decay class 1, 2 or 3 are considered “sound” and assigned a specific gravity of 0.4. Decay class 4 or 5 particles are considered “rotten” and assigned a specific gravity of 0.3. JFiremon also calculates duff and litter volume using bulk densities of 2.75 lbs/ft<sup>3</sup> and 5.5 lbs/ft<sup>3</sup>, respectively (Lutes 2006). The software automatically converts English values into metric, which this analysis reports.

We exported FMAT attribute tables into MS Excel spreadsheets for further analysis. We applied Analyse-It software (<http://www.analyse-it.com>) in order to:

- Verify FMAT outputs.
- Determine normalcy of the dataset.
- Determine confidence intervals around mean values.
- Test for significance using paired t-tests with *P* values Bonferroni adjusted.
- Visually represent sample means, variances, and confidence intervals.

Finally, we subjected statistically non-significant results to post hoc power analysis using NCSS/PASS software (Hintze 2004). We solved for power and  $\beta$  by inputting mean values of each monitoring variable, their standard deviation, the acceptable false change (type-I) error rate ( $\alpha = 0.2$ ), and MDC values specified in Table IV-2.

## Results

### *Forest Structure and Composition*

Before thinning, Douglas-fir was the most abundant tree species in total stems (58%), largely in the form of saplings (2258 stems/ha), although it was present in all size classes (Figure IV-4). The balance of tree stems consisted of Pacific madrone (11%), sugar pine (9%), canyon live oak (9%), ponderosa pine (7%), California black oak (6%), and incense cedar (<1%) (Figure IV-4). Among mature trees >10 cm dbh, Douglas-fir comprised the most basal area (44%) followed by madrone (27%), sugar pine (12%), ponderosa pine (8%), and black oak (6%) (Figure IV-5).

Table IV-6 shows stand structure variables before and after thinning treatment. Thinning significantly increased mean live tree diameter by 24% from 23.9 cm dbh to 29.9 cm dbh ( $p = 0.014$ ) and canopy base height by 40% from 2.2 m to 3.1 m ( $p = 0.004$ ). It also significantly reduced live tree density by 58% ( $p < 0.001$ ) with the greatest effect on saplings (<10 cm dbh) of Douglas-fir (93% reduction) and madrone (94% reduction) (Figure IV-4). However, it decreased mature tree basal area just 14% below the pre-thinning baseline (Figure IV-5). Thinning reduced mature Douglas-fir density by 84% and basal area by 19% by concentrating on smaller trees 10-30 cm dbh. It reduced density of madrone in the same class by 75% and basal area by 23% (Figures IV-4, 5).

After thinning, Douglas-fir was evenly abundant in total stems (22%) with sugar pine (21%) and live oak (20%), although the latter were present only as seedlings. The relative abundance of ponderosa pine (16%) and black oak (14%) also increased above the baseline condition (Figure IV-4). Before and after thinning, conifers dominated the

upper size classes of mature trees (>30 cm dbh) and hardwood species co-dominated with conifers in smaller classes (Figure IV-4).

**Table IV-6.** Means (standard error) of stand structure variables.

	Unit 1		
	<u>Pre-thinning</u>	<u>Post-thinning</u>	<u>% change</u>
<b>Trees &gt;10 cm dbh</b>			
Mean live diameter (cm)	24 (1.9)	<b>29.9</b> (3.2)	24
Mean dead diameter (cm)	26.3 (2.8)	28.2 (3.1)	7
Canopy base height (m)	2.2 (0.1)	<b>3.1</b> (0.1)	40
Tree density (stems·ha) live	400 (92.6)	<b>246.7</b> (64.3)	-38
Tree density (stems·ha) dead	91.7 (18.7)	68.3 (18.5)	-25
Basal area (m <sup>2</sup> ·ha)	26.9 (6.2)	23 (5.7)	-14
Relative density index	0.76	0.53	-30
<b>Trees &lt;10 cm dbh</b>			
Saplings >2.5 cm dbh (stems·ha)	2955 (394.5)	<b>538.3</b> (76.2)	-81
Seedlings <2.5 cm dbh (stems·ha)	1213 (295.5)	1113 (255)	-8
<b>Total stems·ha</b>	4660 (476.2)	<b>1967</b> (236)	-58

**Bold values** indicate statistically significant change ( $p < 0.02$ ).



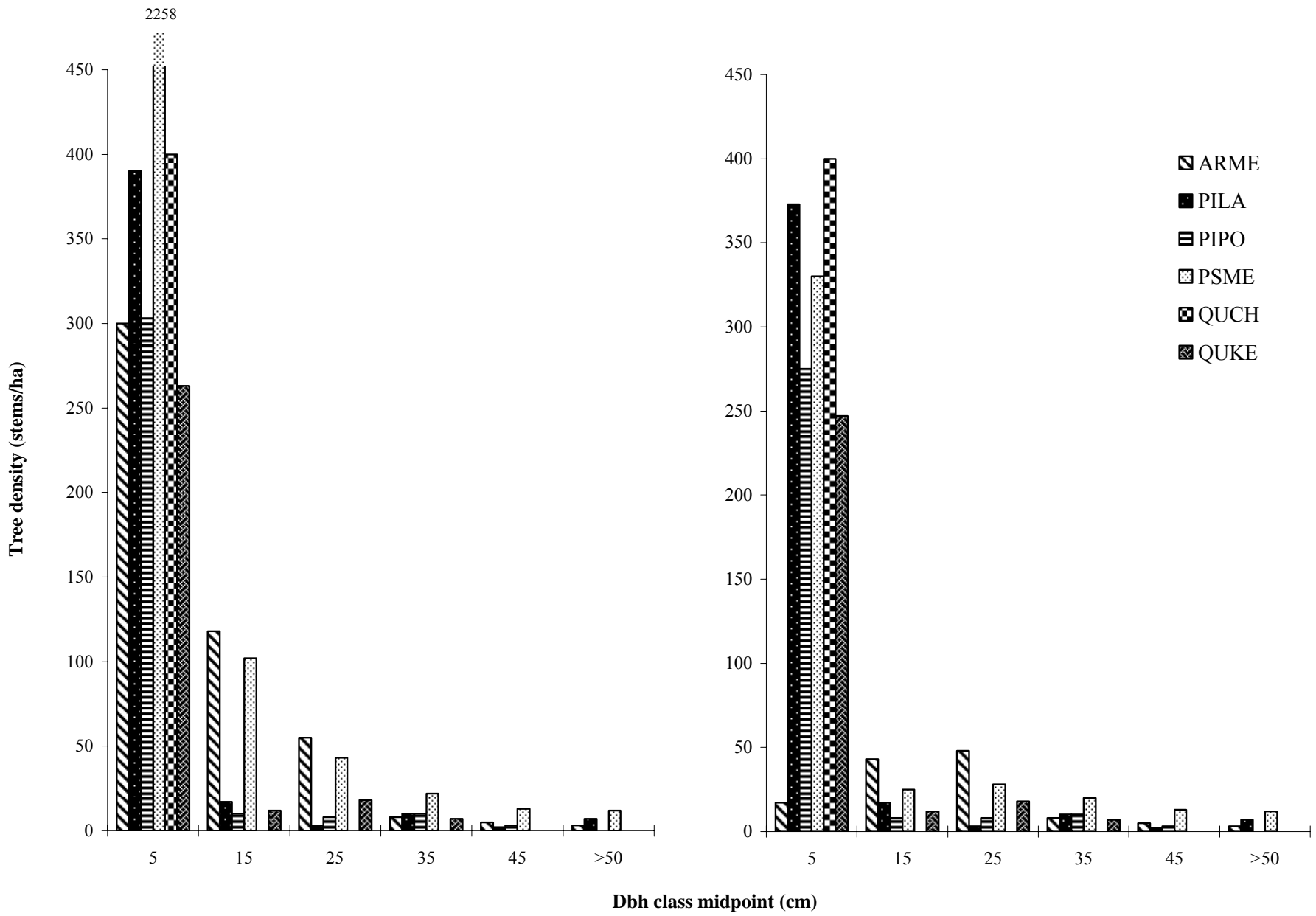
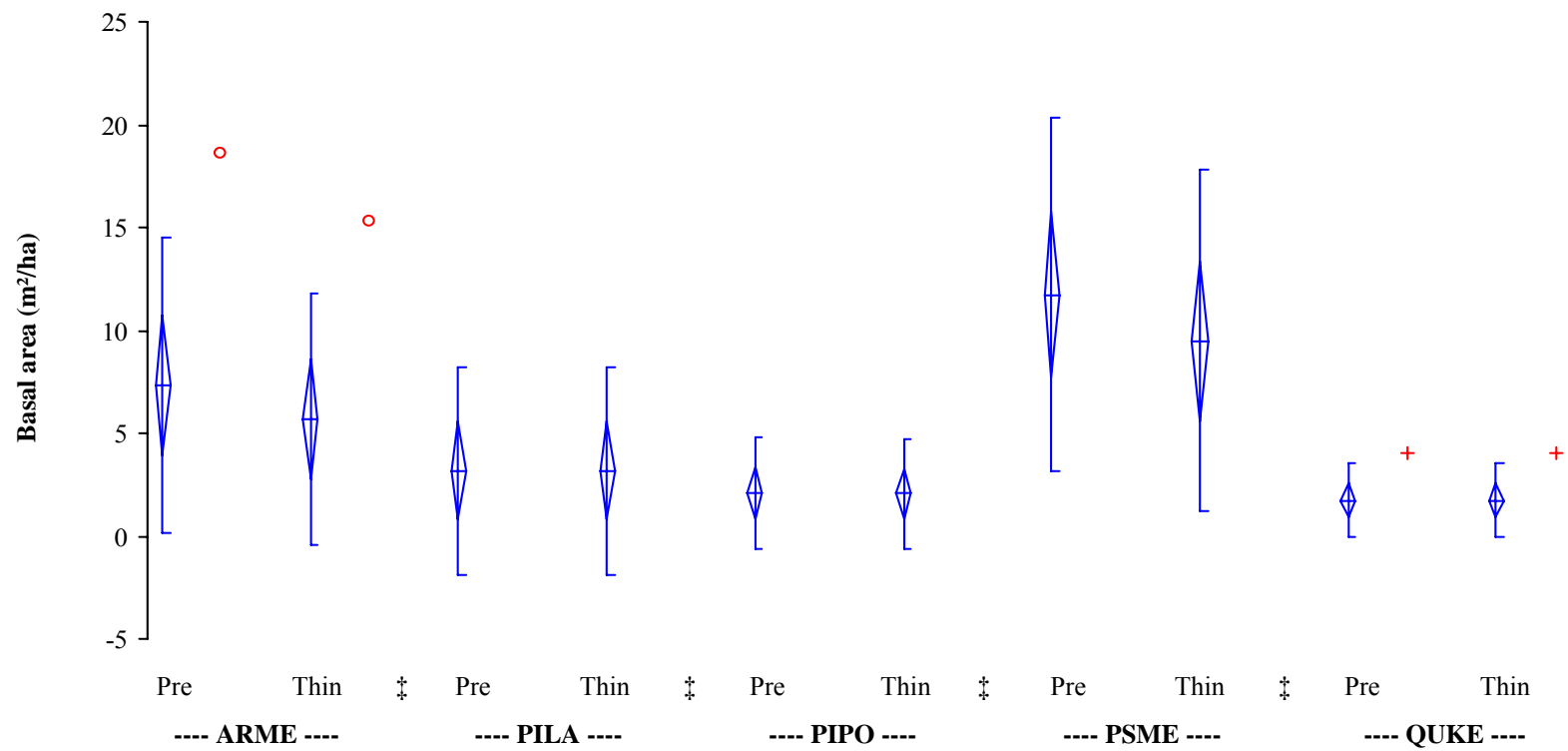


Figure IV-4. Tree density by species and size class before and after thinning in Unit 1.



**Figure IV-5.** Mature tree (>10 cm dbh) basal area by species before (pre) and after thinning (thin).

### *Surface Fuel*

Mean surface fuel load was 7.41 kg/m<sup>2</sup> before thinning. Duff and litter comprised the bulk (65%) of surface fuels, and the remainder consisted of woody debris. Coarse wood particles >7.6 cm in diameter comprised 1.56 kg/m<sup>2</sup> (60%) of surface wood volume, with the balance in finer particles. Among fine wood particles, 100-hour fuels were most voluminous (66%) followed by 10-hour fuels (27%) and 1-hour fuels (7%).

Table IV-7 compares treatment effects on surface fuel variables before thinning, after thinning but before slash treatment, and after pile burning slash treatment. Thinning added 2.01 kg/m<sup>2</sup> of woody debris to the ground, increasing total surface fuel load by 27% ( $p = 0.004$ ) above the pre-treatment baseline, mainly due to a significant ( $p = 0.004$ ) pulse of coarse wood particles in the form of severed tree stems (Figure IV-6). Fine wood load also increased by 54% ( $p = 0.035$ ) after thinning but before slash disposal.

Slash disposal through pile burning reduced total surface fuel load to 7.44 kg/m<sup>2</sup>, or slightly above (<1%) the pre-thinning baseline ( $\beta = 0.196$ ). Woody fuel volume, consisting mainly of coarse particles (1.91 kg/m<sup>2</sup>), remained 17% above the baseline after pile burning (Figure IV-6). Total fine particle volume also was 9% above the baseline. However, among fines, pile burning reduced 100-hour fuels to 10% below baseline levels (Table IV-7). Pile burning also reduced duff (-9%) and litter (-6%) below the baseline, but this was isolated to pile locations and did not occur throughout the treatment area.

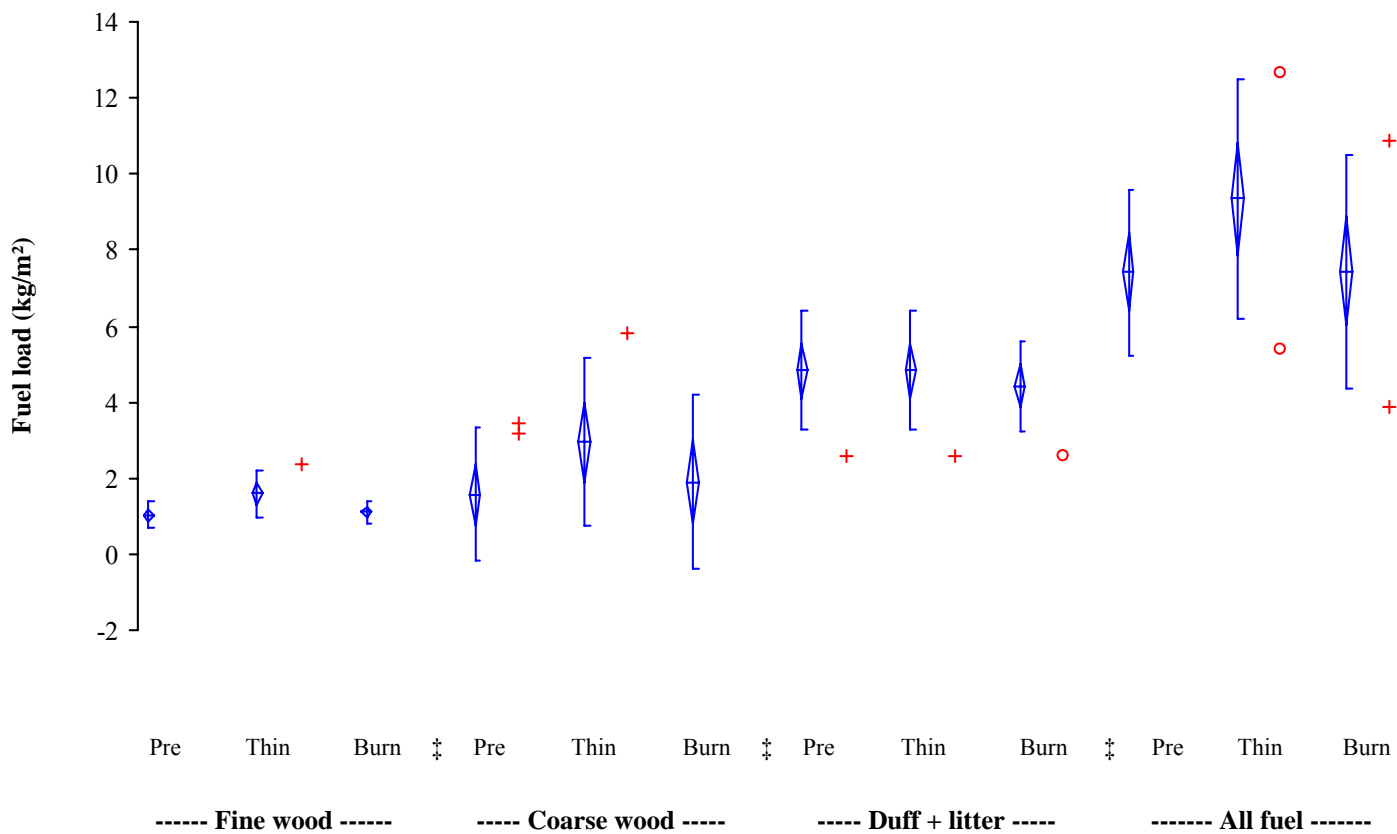
**Table IV-7.** Means (standard error) of surface fuel loading.

	Unit 1				
	<u>Pre-thinning</u>	<u>Thin-only</u>	<u>% change</u>	<u>Thin-and-burn</u>	<u>% change</u>
<b>Fine wood</b>					
1-hour	0.07 (0.01)	<b>0.11</b> (0.01)	57	<b>0.11</b> (0.01)	57
10-hour	0.28 (0.06)	<b>0.58</b> (0.07)	107	0.40 (0.06)	43
100-hour	0.67 (0.08)	0.90 (0.13)	34	0.60 (0.06)	-10
Total fine	1.02 (0.11)	<b>1.59</b> (0.20)	54	1.11 (0.10)	9
<b>Coarse wood</b>					
1000-hour sound	0.13 (0.13)	<b>0.87</b> (0.18)	569	<b>0.50</b> (0.27)	285
1000-hour rotten	1.43 (0.59)	<b>2.14</b> (0.76)	50	1.41 (0.73)	-1
Total coarse	1.56 (0.56)	<b>3.01</b> (0.70)	93	1.91 (0.73)	22
<b>All wood</b>	2.58 (0.59)	<b>4.59</b> (0.86)	78	3.02 (0.80)	17
<b>Duff</b>	3.79 (0.45)	3.79 (0.45)	0	3.45 (0.38)	-9
<b>Litter</b>	1.03 (0.12)	1.03 (0.12)	0	0.97 (0.13)	-6
<b>Total fuel</b>	7.41 (0.69)	<b>9.41</b> (1.00)	27	7.44 (0.97)	<1

All values in in kg/m<sup>2</sup>

1-hr, <0.6 cm. 10-hr, 0.6-2.5 cm. 100-hr, 2.5-7.6 cm. 1000-hr, >7.6 cm.

**Bold values** indicate statistically significant change ( $p < 0.04$ ).



**Figure IV-6.** Surface fuel load by size class before thinning (pre), after thinning (thin), and after slash treatment (burn).

## **Discussion**

### *Forest structure and composition*

Thinning of mostly small trees (<20 cm dbh) significantly reduced stand density from 3443 to 853 stems/ha (Table IV-6). It retained all conifers >50 cm dbh and most hardwoods >20 cm dbh, leaving the dominant and co-dominant trees and snags of every species intact. Thinning also reduced the relative density (RD) of mature trees (0.53) to less than before (0.76), but did not meet the five-year target value (<0.5) (Table IV-1). The post-thinning RD is partly due to the high residual density of live Pacific madrone trees in the 10-30 cm dbh size class (105 stems/ha) (Figure IV-4).

Pacific madrone's drought tolerance, its affinity for maximum sunlight, and its ability to regenerate from seed as well as sprout after top-kill enabled it to successfully compete in the environmental conditions created by historical high-grade logging (Tappeiner et al. 1986). Madrone is widespread in lower elevation forest stands throughout the Applegate River watershed, but its preference for early-successional conditions without much overstory canopy shading (Franklin and Dyrness 1988) and its easily destroyed bark would render it less common in reference stands dominated by large conifers that endure frequent low severity fires (Main and Amaranthus 1996). Additional thinning of madrone stems <30 cm dbh in Unit 1 could hasten development of reference conditions by reducing mid-story canopy bulk density to benefit shade-intolerant black oak, pines, shrubs, and grasses. It also would promote attainment of longer-term stand density management goals (Table IV-1).

Retention of virtually all oaks and pines shifted stand composition toward reference conditions by increasing the relative abundance of those historically dominant

species (Figure IV-4). However, this initial treatment merely nudged stand structure and composition toward the reference rather than imposing it in one action, and thus largely avoided irreversible effects consistent with the prescription intent (Bey 2005). The growth rates of remaining trees may increase as a result of thinning, and tree vigor may be enhanced with less competition for light, water, and nutrients (USDI 1999). Larger trees may develop more quickly in upper canopy layers, increasing the likelihood that large coarse wood will exist over time. These changes favor long-term attainment of reference conditions but follow-up treatments that further reduce stand density and basal area, while also promoting oak and pine, may be necessary.

#### *Surface fuel and potential fire effects*

The results permit a clear distinction between thinning effects on surface fuel load and pile burning effects. The former significantly increased woody surface fuels in all timelag size classes while the latter brought woody fuel load nearer to, but not below, the pre-treatment baseline (Table IV-7, Figure IV-6). After slash treatment, comparison of multiple sampling events did not indicate a significant overall change in surface fuel load, with the sole exception of a significant increase in the volume of 1-hour fuels from 0.07 to 0.11 kg/m<sup>2</sup> ( $p = 0.007$ ). Our failure to detect more change in surface fuel load after slash disposal may represent type-II error, as our sampling design had relatively low power (0.8042), or it could indicate a true absence of post-treatment effect.

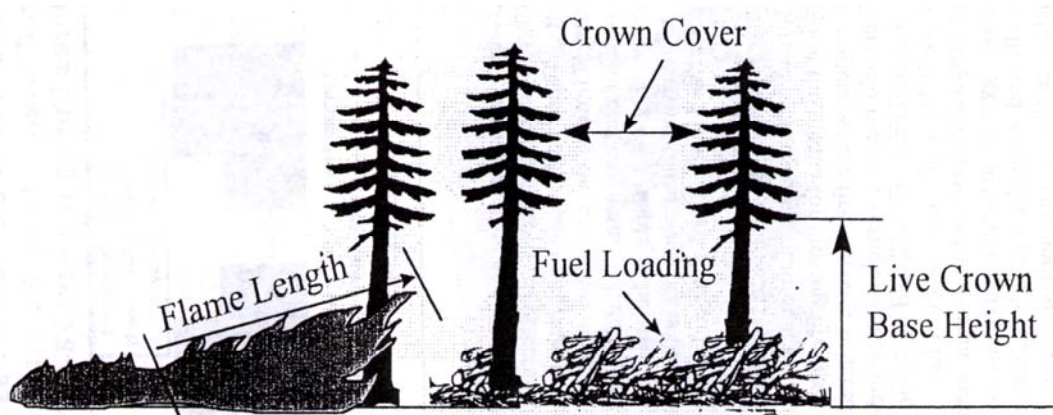
Thinning may have improved stand resilience to fire disturbance by increasing mature live tree mean diameter and canopy base height, in addition to its significant overall reduction of stem density (Table IV-4, Figure IV-4). Fire resistance of Douglas-fir increases with stem size as bark thickens and crown bases rise above the ground (Agee

1993). Trees selected for retention therefore could be more likely to survive heat stress from flaming combustion. Thinning also removed significant portions of sub-canopy Douglas-fir and Pacific madrone that low intensity fire otherwise would have killed (Figure IV-4). Removal of those trees reduced live fuel “ladders” that can facilitate flame movement from the ground surface into the canopies of dominant trees (Agee 1996, Figure IV-7).

Omi and Martinson (2002) study eight areas burned by wildland fire and assess the effects of pre-fire fuel treatments on subsequent fire severity. In that study,

height to live crown, the variable that determines crown fire initiation rather than propagation, had the strongest correlation to fire severity ... [W]e also found the more common stand descriptors of stand density and basal area to be important factors. But especially crucial are variables that determine tree resistance to fire damage, such as diameter and height (22).

The investigators do not characterize surface fuel profiles that existed before the fires subjected to study, and the spatial scale of the events considered confounds replication. However, the authors claim their results can be extrapolated elsewhere. A key implication is that treating live fuels below dominant tree canopies and reducing small-diameter tree



**Figure IV-7.** Forest stand structural elements that influence stand replacing wildland fire effects. Surface fuel loading and vertical height to live tree crowns exert the greatest influence on canopy fire initiation. Source: Agee (1996).



density can reduce the likelihood of stand replacing wildland fire (Figure IV-7). Indeed, research suggests that raising stand-scale canopy base height is a critical factor in disrupting canopy fire initiation (Carey and Schumann 2003, Graham et al. 2004).

As discussed in Chapter III, canopy fire behavior depends on the density of canopy fuels as well as the availability of surface fuels to sustain convective heat transfer into the canopy (Scott and Reinhardt 2001, Figure IV-7). In order for active crown fire to spread independent of surface fireline intensity, rare combinations of topography and weather must be present, and even then, it usually affects only small areas before ceasing on its own (Van Wagner 1977). In Unit 1, flat terrain mitigates the risk of active canopy fire, and fuel treatments that effectively reduce surface fireline intensity can further diminish passive crown fire spread potential (Agee 1996). In contrast, steeper terrain >30% in portions of Unit 2 elevates active canopy fire risk, but that site hosts rare clustered lady's slipper orchid, which associates with closed canopy forest. Canopy bulk density fuel reduction there could trade-off with biodiversity conservation objectives (USDA/USDI 1994b). Furthermore, investigations of fire severity patterns at stand and landscape scales in the KS region correlate low severity fire effects with closed canopy forest structure and high severity effects with conditions created by crown bulk density treatments (Odion et al. 2004, Raymond and Peterson 2005).

Low thinning followed by slash disposal via pile burning facilitated attainment of some, but not all, management goals associated with fire resilience (Table IV-1). The remaining sapling density (538 stems/ha), mature live tree mean diameter, and canopy base height closely resemble the target values (Table IV-6). However, the surface fuel load (7.44 kg/m<sup>2</sup>) exceeds the target value by 24%, and indeed, it is more voluminous

than in the pre-treatment condition (Table IV-7). The amount, continuity, and moisture content of fine woody fuels strongly influence the rate at which fire spreads as well as the intensity of heat energy release at its flaming front (Rothermel 1983). Unless treated, this downed woody fuel burning in a relatively desiccated environment lacking horizontal wind breaks with potentially swifter mid-flame wind speeds may contribute to elevated fireline intensity and resultant severity of fire effects on vegetation and soil than is assumed in a reference condition. Latent duff accumulations, only minimally affected by initial treatments (Table IV-7), also could augment fire severity by prolonging the residence time of smoldering combustion in the soil (Harrington 2000).

BLM assessments conducted in the Applegate River watershed affirm that increased woody surface fuel loads left after mechanical thinning operations produced faster rates of fire spread and longer flame lengths, resulting in intensified fire behavior and increased difficulty of control compared to an untreated condition (e.g., USDI 2005). Raymond and Peterson (2005) similarly observe that elevated woody fuel volumes left on the ground after mechanical thinning contributed to increased fire intensity and stand mortality in treated areas compared to untreated stands of mixed evergreen forest that burned in the 2002 Biscuit fire. Opportunities for further analysis to determine residual fire hazard in the study area and alternative ways to address it include modeling of potential fire behavior and effects using BehavePlus (Andrews 2005), Nexus (Scott 1999), and the Fire and Fuels Extension to the Forest Vegetation Simulator (Ferguson 2004).

### *Methodological efficacy*

The hybridized data collection and analysis protocols used in this study generated useful data for evaluation of treatment effects, although certain advantages and disadvantages of the methods we used merit discussion. One key advantage of the permanent EM sampling plot design following FMH protocols (USDI 2003) was its allowance of dependent-variable statistical testing of data, which requires small sample sizes to accomplish the monitoring objectives (Table IV-2, Figure IV-2). In contrast, the temporary variable-radius plots suggested in FIREMON (Lutes 2006) would necessitate independent-variable testing and require larger sample sizes due to poor data correlation between sampling events (Zar 1996). Permanent plots are most appropriately used where a high degree of correlation links data collected at one location over time, particularly in this study, where monitoring is limited to features that are not prone to spatial movement (Elzinga et al. 1998). Use of permanent EM plots thus supported a core purpose of this monitoring project to maximize the simplicity and efficiency of data collection and analysis.

However, an important disadvantage of permanent plots is the ostensible need they create for two years' worth of pilot data to determine an adequate sample size (Elzinga et al. 1998). The sample size equation we used requires input of standard deviations of the difference between mean values in two sampling events. As an alternative, we determined sample size using standard deviations of only the first-year pilot sample coupled with an arbitrary estimate of the correlation coefficient after the suggestion of Elzinga and colleagues (1998:354). The correlation coefficient we used (0.8) is conservative given that the scope of this analysis is limited to spatially fixed tree

density and fuel load variables. Calculating the sample size in this way maximized the value of first-year pilot data and enhanced project efficiency by sparing the LRP expenses associated with additional data collection and delayed treatments.

Our use of 0.1 ha (1000 m<sup>2</sup>) EM plots also disadvantaged sampling efficiency because the relatively small size of Unit 1 (~16.2 ha) forced us to reject many randomized plot locations and orientations per FMH protocol (USDI 2003). Otherwise, the plots would not have entirely fit within the treatment area. As a result, we randomly established anchor points, and then arbitrarily oriented plots parallel to magnetic north in order to maximize their distance from unit boundaries. Smaller plots can avoid such difficulty in a similarly sized treatment area. The “rapid assessment” plot design of Paintner (2003), for example, could better accommodate random location and orientation while sustaining the benefits of a permanent sampling unit, although its efficacy in measuring forest density and other structural properties is uncertain.

The hybridized protocols introduced two additional inefficiencies that should be noted. First, shortcomings in the FIREMON software forced us to employ additional resources in the sampling design and data analysis stages. In particular, FMAT lacks a means to test statistical power and quantify potential for missed-change (type-II) error. Reporting of non-significant results is incomplete without post hoc power analysis that quantifies the probability of a statistical test failing to detect a true change (Elzinga et al. 1998). Therefore, we used NCSS/PASS (Hintze 2004), an easy-to-use software package specifically developed for power analysis and sample size computation, to overcome this limitation.

Second, FMAT automatically applies a one-way ANOVA procedure to calculate variance for a single variable between two or more sampling events. This procedure tests the alternate hypothesis that at least one sample mean is different from the others. The software thus determines if differences exist among means, as well as which means differ (Lutes 2006). However, repeated measure ANOVA assumes sphericity of correlation among sampling events, which is not ideal for monitoring studies because data correlation may vary across sampling events (Zar 1996). This is particularly true of fuel load data sampled on planar intercept transects because small variations in protocol interpretation among sampling crews can introduce bias and skew the data. Permanent stakes and photo points at both ends of each transect minimize but do not eliminate these sources of bias. Elzinga and colleagues (1998) recommend using paired t-tests to compare data from multiple sampling events in permanent monitoring plots, provided that a Bonferroni adjustment is applied to the threshold *P* value (Glanz 1992), because the test remains valid even if the pair wise correlation is unknown (Snedecor and Cochran 1980). Therefore, we applied Analyse-It (<http://www.analyse-it.com>) to run paired t-tests on the FMAT-generated data attribute tables instead of reporting ANOVA results.

This analysis has implications for development of FEAT-FIREMON Integrated (FFI) (Benson et al. 2006), a consolidated fire effects monitoring and data analysis tool designed as an “advanced alternative” to FIREMON (Lutes 2006) and the Fire Ecology Assessment Tool (FEAT). FMH sampling protocols (USDI 2003) work effectively with FIREMON database (JFiremon) and analysis (FMAT) software, as the latter are flexible to user approaches that differ from underlying technical literature. The degree to which FEAT shares the shortcomings we found in the data analysis capabilities of FIREMON

software is unknown. However, FFI could improve monitoring project efficiency by incorporating means to compute the statistical power of sampling designs as well as apply appropriate tests for significance to data sampled from either permanent or temporary plots.

## V. SYNTHESIS

Reference analysis (Chapter III) explains the significance of wildland fire to the adapted disturbance ecology of mixed evergreen forests in the Siskiyou Mountains. It hypothesizes that the study area historically experienced low and moderate severity fire effects associated with anthropogenic burning oriented to the production of food and other essential items of material culture. Noting the importance of ongoing climate change to region-scale fire regimes and its implications for restoration of fire-adapted forests, the analysis suggests reference conditions for ecological restoration in the study area comprising a range of potential stand structures and compositions that promote ecosystem resilience to inevitable fire disturbance. Perpetual exclusion of fire in the study area is neither desirable nor feasible, as it entrenches forest health decline and risks catastrophic loss of property and ecosystem function (USDI 1996, 1999, 2000).

Monitoring of initial treatment activities in Unit 1 (Chapter IV) demonstrates that mechanical thinning of small trees followed by slash disposal through pile burning nudged stand structure and composition toward reference conditions and somewhat improved stand resilience to fire. However, the treatments left significantly more mid-sized stems of Pacific madrone (Figure IV-4) than would be present in a reference condition, and did not reduce pre-existing surface fuels (Table IV-7) that contribute to fireline intensity and severe fire effects (Agee 1996). Therefore, initial treatments did not accomplish all of the short-term management goals for the Penny Stew project, and further action including application of management-ignited prescribed fire (MIPF) is necessary to attain reference conditions in the study area, as discussed below.

### *Fire hazard reduction demands fire use*

Mechanical thinning is more widely used, but management-ignited prescribed fire (MIPF) is the most effective means to calm subsequent fire behavior, particularly where activity fuels remain on the ground after thinning operations are completed (USDI 1998). Stephens (1998) compares 12 different fuel treatment combinations and concludes that MIPF used alone or in combination with low mechanical thinning is the most effective way to minimize fireline intensity in FARSITE-simulated wildfires. MIPF is so effective because flaming combustion consumes fine woody fuels, which offers unique advantages over other treatment methods that focus on larger, less flammable fuels (Deeming 1990). Indeed, van Wagendonk (1996) compares effects of alternative fuel treatments and reports that low mechanical thinning and pile-and-burn slash treatments on flat ground yield similar fire behavior to low thinning without any slash treatment (i.e., more intense than no treatment) because pre-existing surface fuels remain unaffected. Therefore, effective fire hazard reduction in the study area may depend on the rate at which surface fuels are treated with MIPF.

### *Ecological restoration demands fire process*

Some researchers associate forest structural patterns with various fire regimes and recommend mechanical treatments to “mimic” effects of low- and moderate-severity fires (Arno and Fiedler 2005, Weatherspoon 1996). However, manipulations of forest structure alone do not replicate fire disturbance, particularly its thermal effects on seed germination and nutrient cycling (Agee 1993). Ecological restoration in the study area requires establishment of not only the reference forest structure and composition, but also



the disturbance processes needed to sustain reference structure and composition over time (McIver and Starr 2001). Reliance on structural manipulations of forest vegetation and continued exclusion of wildland fire may perpetuate ecosystem degradation (Atzet 1996).

#### *Spatial pattern of fire use influences effects*

Even if land managers use MIPF in the study area, its scale of use will fall short of restoring the functional role of fire process on the landscape (Martin 1997). MIPF operations usually are kept small to ensure control and generally are not done with landscape-scale restoration objectives in mind. Baker (1994) asserts that small-scale MIPF operations can intensify adverse cumulative effects on landscape structure caused by fire exclusion. Indeed, uncontrolled wildland fires offer distinct advantages for the restoration of landscape structures altered by fire suppression because no other option mimics the spatial patterning of historical fire effects (Baker 1989). Fire should be reintroduced at a landscape scale, thereby allowing natural ecological processes to shape ecosystem structure and composition over time (Hardy and Arno 1996). The most appropriate places to implement landscape-scale fire restoration include roadless areas and large blocks of lightly roaded areas where risks to human life and property are low (DellaSala and Frost 2001). In contrast, a sensible approach to stand-scale forest restoration in the study area, a rural-wildland interface zone, would prepare the site to support low severity wildland fire with applications of MIPF.

#### *Timing of fire use influences effects*

Use of MIPF can restore some ecosystem processes that have been limited or rendered dormant by fire suppression (Arno 2000). MIPF has been used effectively in the

restoration and maintenance of fire-adapted plant communities and wildlife habitat (Hardy and Arno 1996, McMahon and deCalesta 1990). However, as currently practiced, it probably cannot mimic all of the ecological functions of historical fires. Land managers in the Siskiyou typically ignite prescribed fires during the wet season to minimize smoke production and risk of escape (Chandler 2002). Wet season burning may negatively affect soil microorganisms by more efficiently conducting heat deeper into soil layers than would occur in the dry season (Agee 1993), although no experimental or monitoring data supports this speculation (Ryan 2003).

Research shows that little soil heating occurs if soils are very moist at the time of burning and fireline intensity is low (DeBano et al. 1979). In this situation, hard-seeded chaparral plants may not receive sufficient heat shock to germinate, resulting in high seed mortality and low seedling emergence (Odion 2002). Thus, non-sprouting shrubs (e.g., *Arctostaphylos viscida*, *Ceanothus integerrimus*) may experience poor regeneration and seed bank depletion as a result of wet season burning. Those species are not likely to be resilient to such fire effects (Odion 2002). Burned areas with poor shrub regeneration are more prone to invasion by yellow star thistle (*Centaurea solstitialis*) and cheat grass (*Bromus tectorum*), among other exotic weeds and annual grasses. Fine fuels made by exotics can easily carry fire and enable reburns before shrubs can produce seed (Odion 2002). In the worst-case, site conversion to a short-rotation (<5 years) fire regime fueled by exotic plants can eliminate native chaparral (Zedler et al. 1983).

#### *Guidelines for use of fire in the study area*

As noted in the reference analysis (Chapter III), passive restoration, defined as cessation of practices that degrade ecological function (i.e., fire suppression) may be an

appropriate restoration paradigm for Unit 2 of the study area (DellaSala et al. 2004). However, if active management is considered, MIPF applications could be justified in the oak savannah near the unit center, the pine flats on its east flank, and at lower slope positions occupied by mature Douglas-fir where the terrain is relatively flat and soils are stable. In areas where fuel loading is too high to safely apply MIPF, limited mechanical thinning may help to actualize desired forest structure, and in turn hasten the use of MIPF in ways that more closely emulate historical ecological functions of fire (Brown 2000b). Manual fuel reduction treatments including pile-and-burn and ladder fuel pruning can effectively clear fine surface and ladder fuels and prepare a site for burning without significantly impacting soils or wildlife habitat (Graham et al. 2004). Such pre-treatments already have been accomplished in Unit 1.

Successful use of MIPF may require sequential applications before reference conditions are realized. Knowledgeable workers (e.g., Agee et al. 2000, Martinez 2004) recommend staggering burn treatments by several years because excessive frequency can deplete seed banks (discussed above) and simplify multi-layered habitats used by sensitive wildlife such as northern goshawk (DellaSala et al. 1995). Burns should be accomplished under conditions in which important structural elements of habitat, such as large and old trees and coarse wood, can be protected (Agee and Huff 1986). As always, periodic monitoring should follow MIPF treatments to determine whether restoration goals have been met and when such activities should cease.

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