DEVELOPING FIRE MANAGEMENT STRATEGIES IN SUPPORT OF ADAPTIVE MANAGEMENT AT TEJON RANCH, CA

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April 2012

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The mission of the Bren School of Environmental Science & Management is to produce professionals with unrivaled training in environmental science and management who will devote their unique skills to the diagnosis, assessment, mitigation, prevention, and remedy of the environmental problems of today and the future. A guiding principle of the School is that the analysis of environmental problems requires quantitative training in more than one discipline and an awareness of the physical, biological, social, political, and economic consequences that arise from scientific or technological decisions.

The Group Project is required of all students in the Master's of Environmental Science and Management (MESM) Program. It is a three-quarter activity in which small groups of students conduct focused, interdisciplinary research on the scientific, management, and policy dimensions of a specific environmental issue. This Final Group Project Report is authored by MESM students and has been reviewed and approved by:

FRANK DAVIS, PH.D.
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ABSTRACT

The purpose of this Group Project is to contribute to the Tejon Ranch Conservancy's Ranchwide Management Plan by 1) investigating the role of fire in the major ecological communities of Tejon Ranch, and 2) developing and evaluating a suite of fire management strategies for these communities. To understand the Ranch's past and present fire regimes, we combined an analysis of the fire record, fire return interval departure mapping, and community-specific research into fire ecology. We then investigated management options, and developed strategies for particular ecological communities. We used LANDIS-II, a spatially-explicit forest landscape model, to simulate the possible effects of climate change, land use change, and management strategies on portions of the Ranch. Finally, we conducted a cost analysis of selected strategies, including different fuel treatments in conifer systems. Based on this work, we recommend the following: 1) continuing fire suppression throughout the Ranch; 2) monitoring the effects of fire suppression and grazing in grasslands and oak woodlands; 3) restoring riparian areas; 4) surveying forest structure and fuel loads of conifer stands to assess the need for potential fuel treatments; and 5) carefully monitoring ground cover of invasive annual grasses in Joshua tree woodlands and desert scrub communities.

I. EXECUTIVE SUMMARY

Comprising nearly 270,000 acres at the confluence of four major ecological regions, Tejon Ranch is the largest contiguous piece of private land in California. Under the 2008 Tejon Ranch Conservation and Land Use Agreement, the vast majority of this land—178,000 acres, with an already-exercised option for 62,000 additional acres—is to be set aside for permanent conservation. The remainder will be divided into three major developments on the Ranch's southwestern side: 1) Grapevine; 2) Tejon Mountain Village; and 3) Centennial. The Agreement is structured so that acreage will be turned over to the independent, nonprofit Tejon Ranch Conservancy as the development process moves forward. In accordance with its mission to "preserve, enhance, and restore the native biodiversity and ecosystem values of the Ranch and Tehachapi Range for the benefit of California's future generations," the Conservancy will prepare a Ranchwide Management Plan (RWMP) by 2013.

This Report is designed to contribute to the RWMP by developing and evaluating a suite of fire management strategies for the Ranch's major ecological communities. It begins by investigating the Ranch's past and present fire regimes. Analysis of the Ranch's fire record reveals that, compared with the period from 1950-1979, fires recorded since 1980 occur more frequently, are larger, are not confined to the traditional fire season, and are largely anthropogenic in origin. Fire return interval departure mapping suggests that these changes are not evenly distributed across the Ranch, and indicates that portions of the Ranch are actually burning less frequently than they did historically. In-depth research into the fire ecology of different communities reveals a complex array of fire regimes, and provides a foundation for understanding how fire management strategies can be used to maintain biodiversity, resilience and human safety.

The Report then discusses the advantages and disadvantages of a range of management strategies. Simulations conducted using LANDIS-II, a spatially-explicit forest landscape model, reveal how selected management actions may interact with succession, disturbance and climate change over a 32,606-acre area that includes most of the Ranch's conifer forests. A cost analysis provides the Conservancy with comparative cost estimates of each management option. Finally, we offer specific management recommendations for the Ranch's major ecological communities, including the following: 1) continuing fire suppression throughout the Ranch; 2) monitoring the effects of fire suppression and grazing in grasslands and oak woodlands; 3) restoring riparian areas; 4) surveying forest structure and fuel loads of conifer stands to assess the need for potential fuel treatments; and 5) carefully monitoring ground cover of invasive annual grasses in Joshua tree woodlands and desert scrub communities.

II. PROJECT SIGNIFICANCE

Tejon Ranch's size, location, and lack of fragmentation provide the opportunity to protect an unusually wide array of species and regional habitats within a single reserve (White et al., 2003). Its landscapes (see Figure 1), which range from grasslands, riparian areas and oak woodlands to montane conifer forests, chaparral communities and deserts, support 61% of the different vegetation communities found in the greater 6.5-million-acre region (Appelbaum et al., 2010). As part of the California Floristic Province—a region identified as a biodiversity hotspot by Conservation International—the Ranch exhibits a high degree of species richness and endemism. It hosts 61 sensitive species and at least 20 species listed as threatened or endangered, including the California condor, the burrowing owl, the Valley elderberry longhorn beetle, and the Tehachapi slender salamander (Dudek, 2009a; White et al., 2003). Moreover, because the transverse Tehachapi Mountains provide an uninterrupted corridor for wildlife travel between the Sierra Nevada and Transverse Ranges, the Ranch links reserves as distant as Sequoia National Forest, Los Padres National Forest, and The Wildlands Conservancy's Wind Wolves Preserve. These linkages are important for ensuring the long-term sustainability of regional ecosystems and protected areas (Tejon Ranch Company & Tejon Ranch Conservancy, 2009; White et al., 2006).

As the Conservancy begins managing this critically important habitat, one of the central questions it will face is how to manage fire in a way that sustains the ecological functioning of the Ranch while promoting human safety. Our Group Project seeks to offer a scientifically rigorous answer to this question. By building a foundation for the adaptive management of fire regimes across the Ranch, our work can assist the Conservancy in preparing the RWMP and in making ongoing fire management decisions.

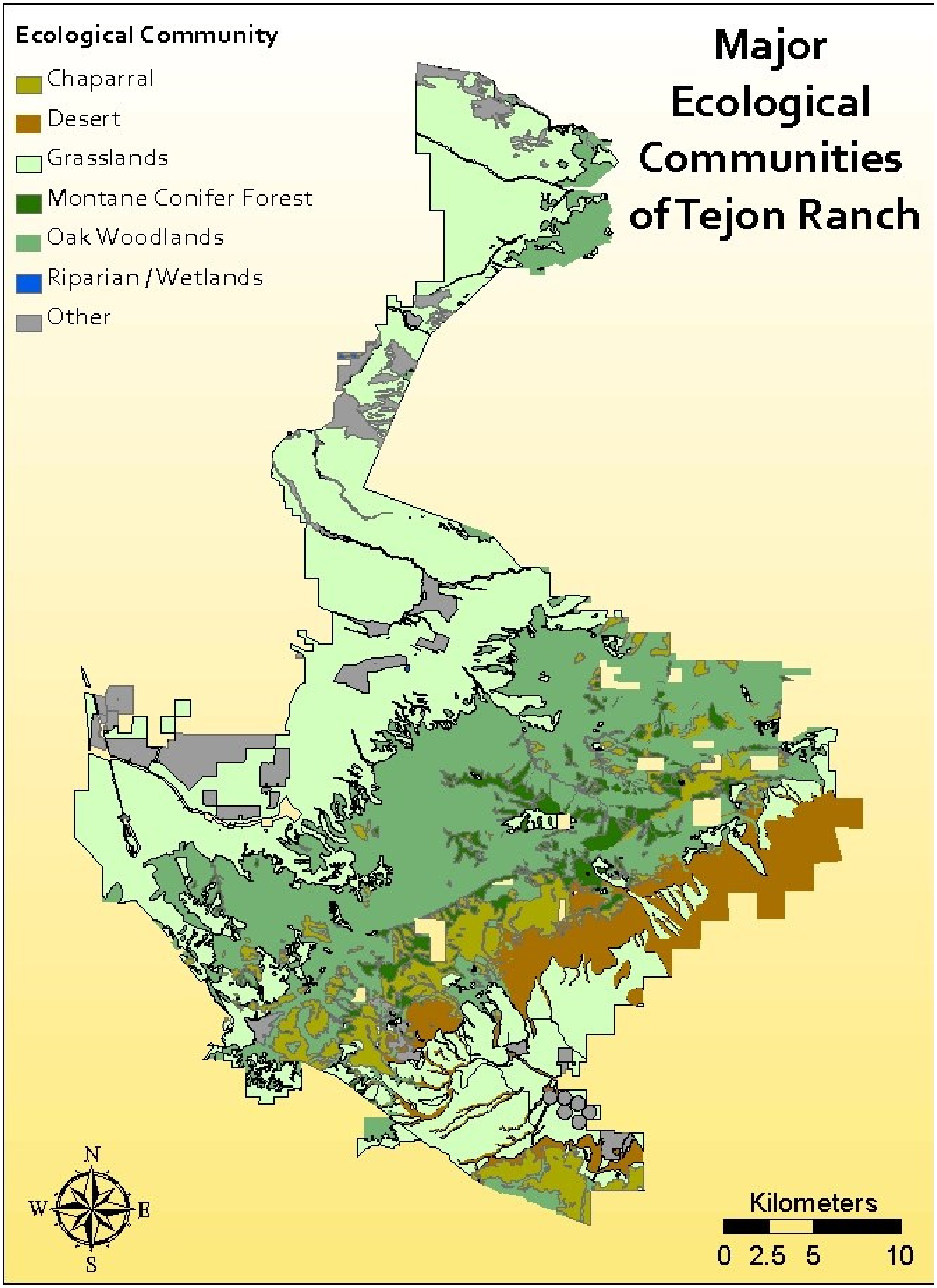

Figure 1: Location and major vegetation types of Tejon Ranch.

III. OBJECTIVES

Our primary objectives were 1) to investigate the role of fire in the Ranch's ecological communities, and 2) to develop and evaluate a suite of fire management strategies to inform preparation of the RWMP. This entailed the following steps:

- Examining frequency, size, seasonality and ignition sources of historical fires on the Ranch;
- Using fire return interval departure analysis to determine post-European-settlement changes to fire regimes;
- Researching past and present fire regimes in each of the Ranch's major ecological communities;
- Researching drivers of fire regimes in each ecological community and across the Ranch as a whole;
- Developing ecosystem-specific fire management strategies;
- Using LANDIS-II to model the impact of alternative fire management strategies on selected vegetation communities in the context of land use change and climate change;
- Determining the impact of fire on focal wildlife species;
- Analyzing the costs of selected management strategies;
- Making fire management recommendations for the Ranch's major ecological communities; and
- Defining key uncertainties and areas for future research.

IV. PAST AND PRESENT FIRE REGIMES

A. FIRE HISTORY TRENDS

In order to quantify the Ranch's fire history, we analyzed fire perimeter data from the California Department of Forestry and Fire Protection (CAL FIRE) dating back to 1950 (see Figure 2). Given the relatively few fires that have occurred on the Ranch since 1950, it is not possible to confirm that any of our observations reflect systematic trends. For this reason, we compared all recorded fires greater than 10 acres in size on the Ranch with fires that occurred in the broader region, defined by the four U.S. Forest Service (USFS) ecoregion subsections that converge on the Ranch (see Figure 3).

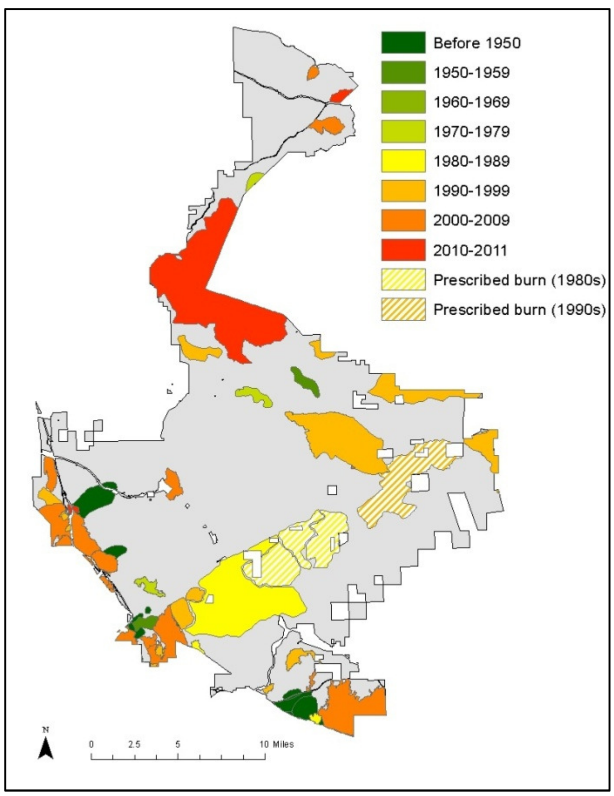

Figure 2: Recorded fires on Tejon Ranch. More recent fires obscure older fires. Only fires occurring between 1950 and 2010 were used in the analysis.

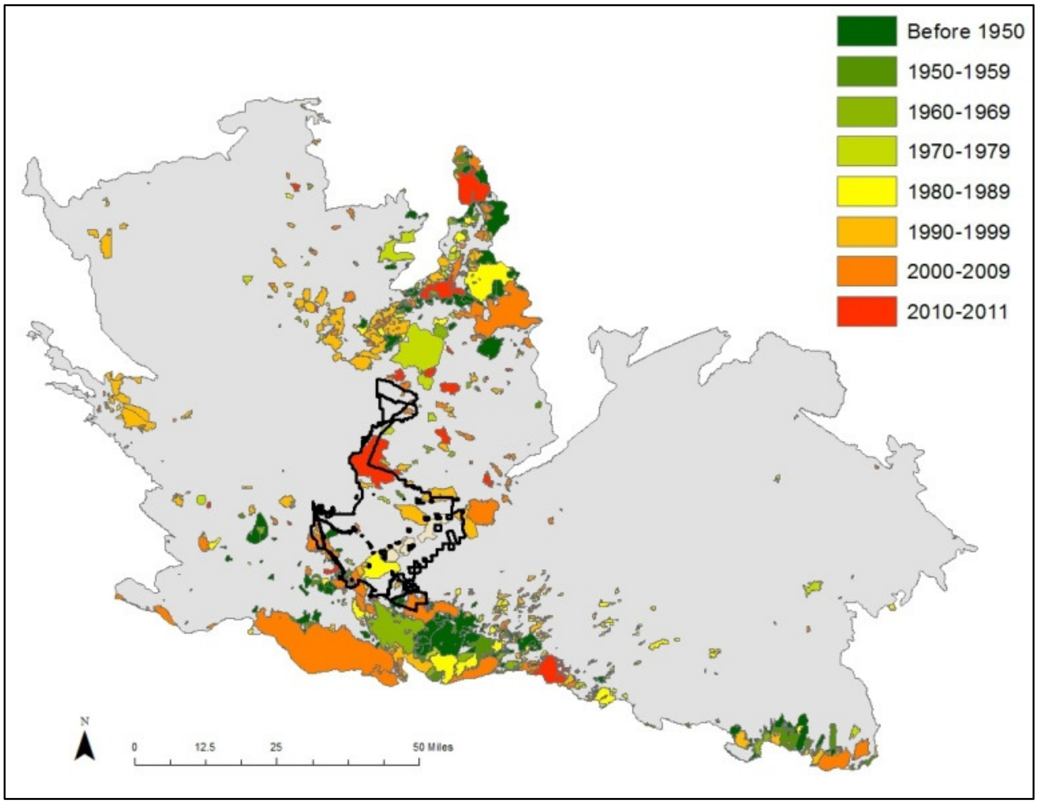

Figure 3: Recorded fires in USFS ecoregion subsections which converge on Tejon Ranch. More recent fires obscure older fires. Only fires occurring between 1950 and 2010 were used in the analysis.

This analysis built on findings by Appelbaum et al. (2010), who determined that 1) fires on the Ranch since 1980 have been both larger and more frequent than fires from 1950 to 1979, 2) the majority of the Ranch has not burned since 1950, 3) the majority of recorded fires have begun during June or July, and 4) significantly more fires have ignited within 1,000 meters of major roads than at further distances.

Our results provide additional support for the finding that fires have been more frequent since 1980 than they were before 1980. We found a statistically significant relationship between the number of fires in the region and time since 1950 (see Figure 4). While this relationship cannot be used to predict future fire frequency, it describes the approximate rate at which fire frequency has been increasing in the region during the period of record.

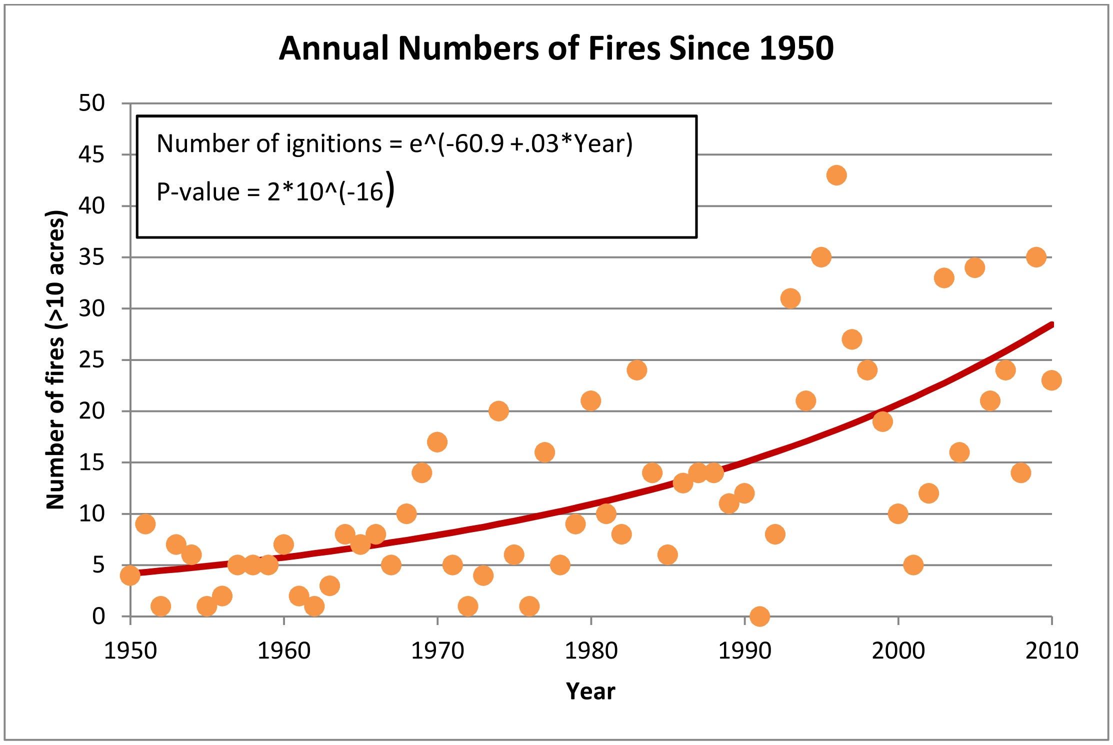

Figure 4: Number of fires each year since 1950. A Poisson regression was used to represent the relationship between number of fires and year.

We also compared average fire size, along with total area burned before and after 1980, on the Ranch and in the broader region. On the Ranch, we found that the average fire size was approximately two times larger since 1980 than in the period between 1950 and 1979. This result is inconsistent with trends in the broader region, where average fire size has been somewhat smaller since 1980.

A comparison of average fire size by season revealed that fires were larger in the fall compared to the summer, likely due to the occurrence of dry fuels and seasonal foehn winds in the fall (see Figure 5). On the Ranch, the average fire size was approximately 1.8 times larger in the fall than in the summer. In the broader region, fall fires were approximately 1.6 times larger than summer fires. It also appears that more fires have been recorded outside the Ranch's traditional fire season in recent decades, although this observation is based on only a few events.

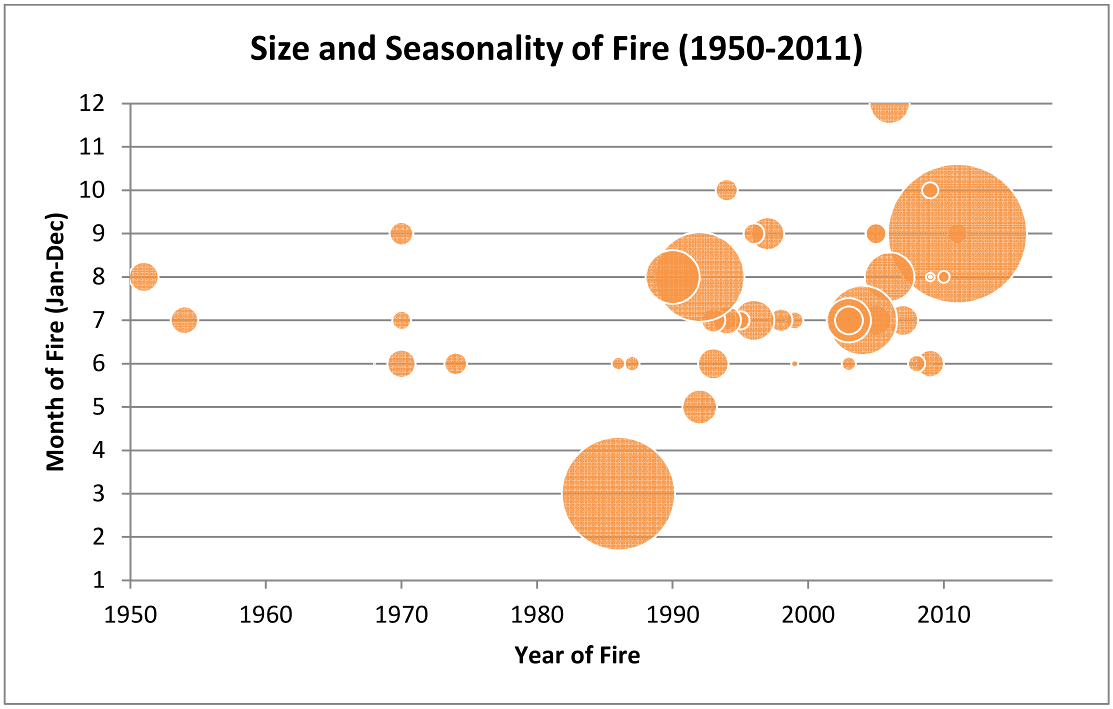

Figure 5: Size and season of fires occurring on Tejon Ranch since 1950 (excludes prescribed burn treatments). The size of each circle reflects the relative area burned. The largest fire represented is 17,644 acres. Only fires occurring between 1950 and 2010 were used in this analysis.

To help explain these patterns, we investigated ignition sources (see Figure 6). We found the three most common to be 1) unknown/miscellaneous, 2) vehicles, and 3) lightning. Among ignitions for which the causes are known, the vast majority are anthropogenic. Only six are due to lightning strikes. The trends observed on the Ranch, including an increase in fire frequency, an increase in average fire size, and potentially an extended fire season, are likely related to the increasing number of anthropogenic ignitions. Climate change and possibly incomplete fire perimeter records for the earlier period may also be influencing these results.

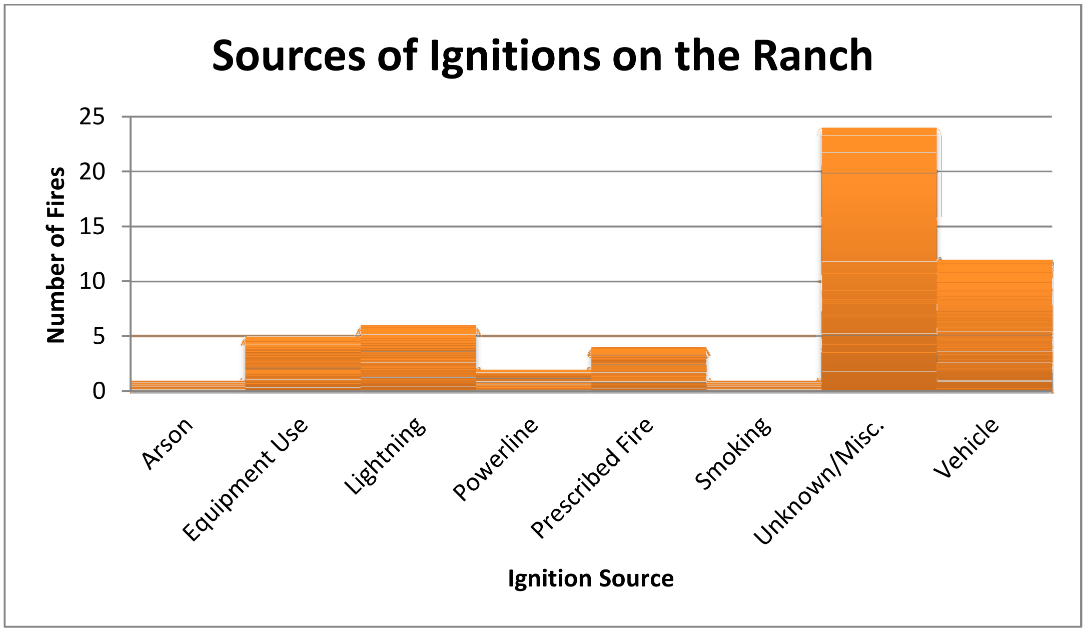

Figure 6: Sources of fire ignitions on Tejon Ranch from 1950 to 2011, derived from CAL FIRE data and Tejon Ranch prescribed fire records. Sources of the 2011 Comanche and Keene fires were obtained from news reports ("4p.m. Update," 2011).

B. FIRE RETURN INTERVAL DEPARTURE ANALYSIS

In order to understand how the Ranch's fire regimes have changed in the centuries since European settlement, we conducted a fire return interval departure (FRID) analysis. This type of analysis is designed to quantify departures from historical fire return intervals (FRIs) in a spatially explicit way (Van Wagtendonk et al., 2002). While it is limited by the accuracy and availability of historical data, and cannot be used as the sole criterion for management decisions in highly altered ecosystems such as invasive-dominated[1] grasslands, FRID provides a useful perspective on how fire regimes in particular ecological communities are changing over time. This analysis can be used to identify areas where fuel loads, vegetation structure, and species composition may be significantly outside historical norms.

1. Methods

The first step in our analysis was to identify historical FRIs for each relevant vegetation type[2] on the Tejon Ranch Company Vegetation Map.[3] For every cover type except grasslands, historical FRIs were obtained from a FRID analysis of California's national forests conducted by the USFS and The Nature Conservancy. This analysis used scientific literature and expert opinion to estimate "[p]resettlement fire regimes" for the 300 to 400 years prior to European settlement, and assigned the same FRI values to CALVEG vegetation types that exhibited comparable historical fire regimes (Safford et al., 2011). FRI estimates for grasslands were obtained from Stephens et al. (2007), who derived their FRI values from accounts of Native

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[1] We use the term "invasive" to refer to nonnative species with unwanted ecosystem effects per Executive Order 13112 and California law (Cal. Food & Agric. Code § 5260.5). Both invasive species and the management of invasive species can drive fire regimes by changing fuel characteristics. Fire, in turn, creates disturbances that can promote certain invasives. Invasive species are therefore relevant to fire management in a number of the Ranch's ecological communities.

[2] Due to lack of data, or lack of recurring fire, the following vegetation types were not included in the analysis: cottonwood/willow riparian; desert wash/riparian/seeps; eucalyptus; oak riparian; riparian scrub; riparian woodland; riparian/wetland; wash; and wetland. We also excluded agricultural land, developed areas, and the proposed developments at Grapevine, Tejon Mountain Village and Centennial.

[3] The Tejon Ranch Company Vegetation Map was compiled from numerous sources, including the 1980 Environmental Impact Report, the 1980 Timber Survey, and the 2003 Comanche Point Vegetation Survey. The vegetation was classified to the Holland system but is not always consistent.

American burning. Because of the influence of extreme values on mean estimates, Safford et al. (2011) described FRIs in terms of median and maximum intervals, a convention that we followed in our own analysis.[4] After obtaining historical FRIs, we used the CALVEG types[5] listed in Safford et al. (2011) and the grassland systems discussed in Stephens et al. (2007) to assign each vegetation type on the Tejon Ranch Company Vegetation Map to a fire regime group. FRIs and sources are listed in Appendix C.

The next step was to identify modern FRIs, which FRID analysis defines as the time since a given area last burned. To do this, we used the Ranch-specific fire records identified in Appendix C. For areas that had not burned since recordkeeping began in 1878, the last burn was assumed to be 1878 (Van Wagtendonk et al., 2002).

We then built and ran two FRID models—one for maximum values, and one for median values—in ArcGIS. For each model, we divided the relevant portions of the Ranch into a raster of 10 m^2 cells, and used Spatial Analyst software to calculate the following equation for each cell:

$$FRID = |[FRI - (2012 - \text{last year burned})]| / FRI,$$

where FRI = historical fire return interval (Van Wagtendonk et al., 2002).

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[4] FRID analysis uses median values, rather than low values, because low values present an unrealistically extreme view of departures from FRIs (Van Wagtendonk et al., 2002).

[5] In some instances, expert opinion rather than the CALVEG classifications was used to place Ranch vegetation types into fire regime groups. For example, we used the ponderosa/white pine FRI for the Ranch's conifer/mixed oak, intermixed conifer, white fir stand, and white fir/mixed oak groups. Though white fir is also found within moist mixed conifer sites in California, the Tejon mixed conifer forests are characteristically drier communities (M. White, personal communication, November 4, 2011).

2. Results and Ranchwide Analysis

Running both models yielded a FRID map for maximum values (see Figure 7), and a FRID map for median values (see Figure 8). The FRID map for maximum values represents a more conservative estimate than the map for median values. Departures shown on the maps represent only areas where the Ranch is burning *less* frequently than the historical maximum or median.[6]

[6] These results are not directly comparable to FRID maps generated by the USFS. The USFS and The Nature Conservancy use a different approach to calculate FRID, and their maps incorporate areas with increased fire frequency relative to historical FRIs (Safford et al., 2011).

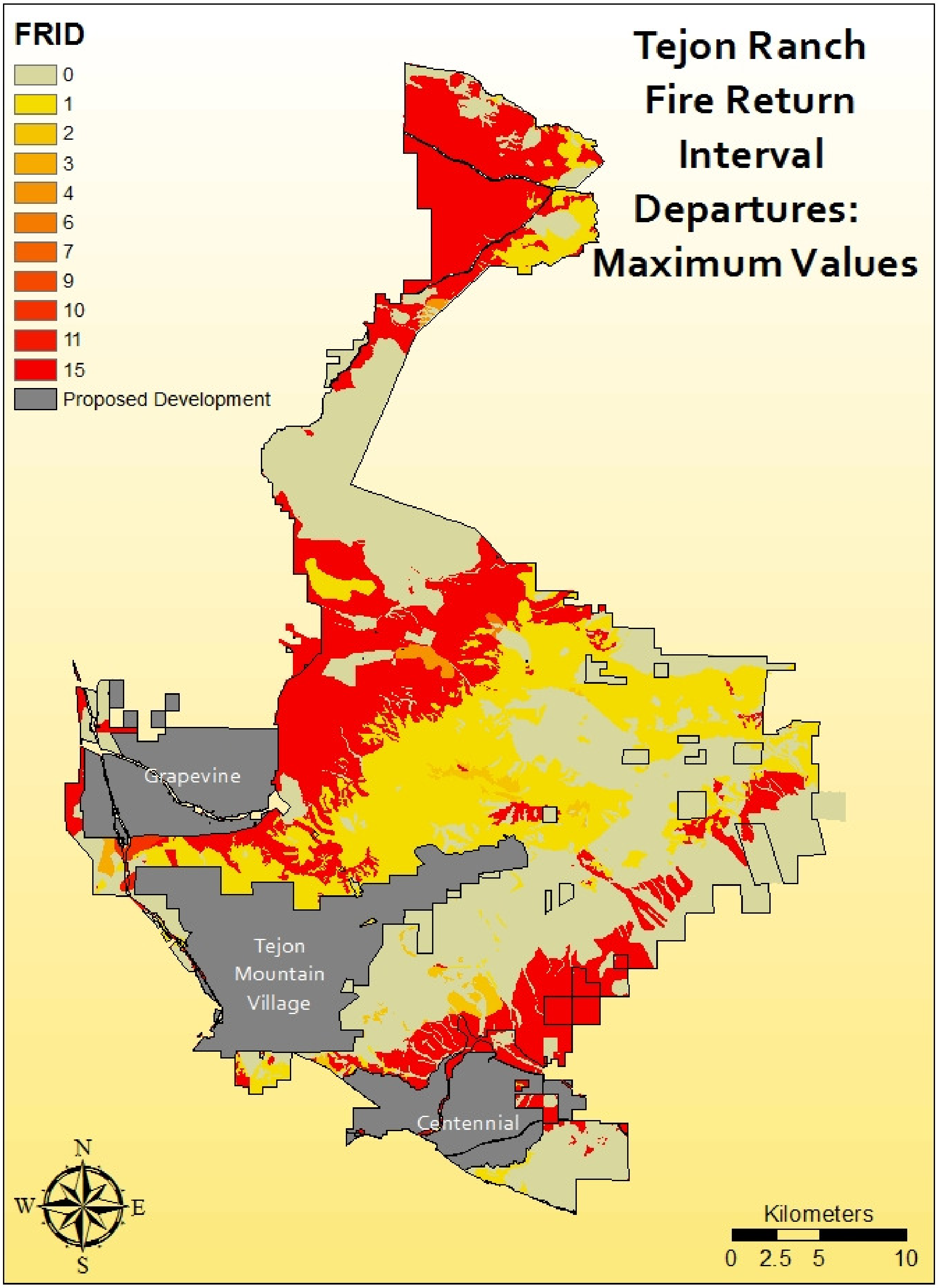

Figure 7: Tejon Ranch FRID map with maximum values. FRID is expressed as the number of intervals since the last fire.

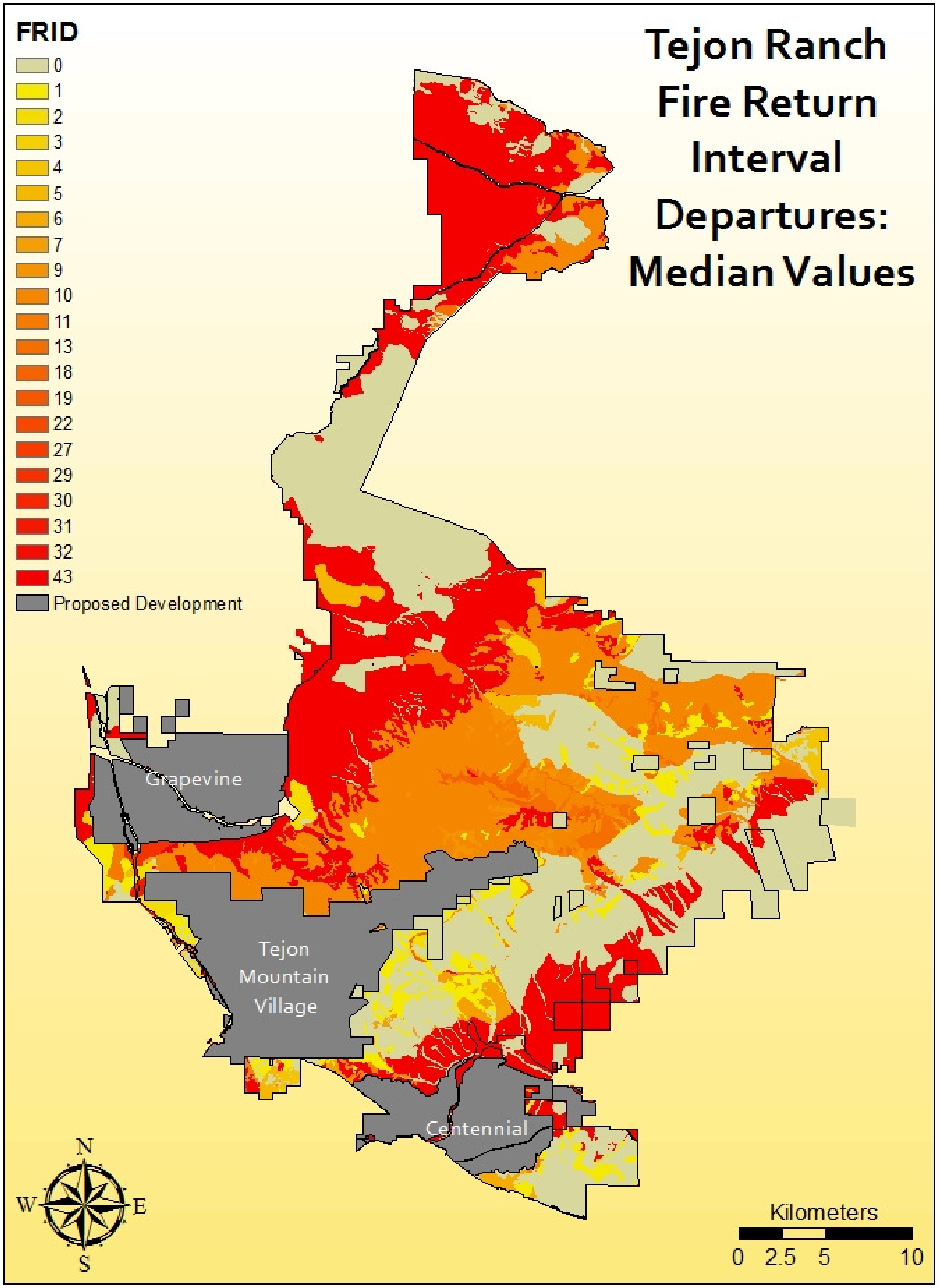

Figure 8: Tejon Ranch FRID map with median values. FRID is expressed as the number of intervals since the last fire.

As these maps suggest, portions of the Ranch have departed significantly from presettlement FRIs, and several high-departure areas are adjacent to proposed developments.[7] Substantial stretches of San Joaquin Valley and Mojave grasslands[8] have no recorded fire history since 1878, and could be as much as 43 FRIs away from estimated presettlement fire regimes. Unburned grasslands dominate the northern tip of the Ranch, border the proposed Grapevine development on three sides, and form much of the northern boundary of Centennial. Elsewhere on the Ranch, the oak woodlands and savannahs to the north and northeast of Tejon Mountain Village show moderate divergence from historical fire regimes, with departures of up to 11 FRIs. While these results do not necessarily establish the need for active management strategies such as prescribed burning, they do suggest the importance of continued monitoring of fuel levels, stand densities, and species composition.

C. FIRE REGIMES IN THE RANCH'S MAJOR ECOLOGICAL COMMUNITIES

1. Grasslands

Located in both the San Joaquin Valley and Antelope Valley, and extending into oak savannahs, grasslands are among the largest of the Ranch's ecological communities. Promoted by the characteristics of grassland fuels, fire has a significant effect on ecosystem functioning (Vogl, 1979). Neither the pre-European fire regime

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[7] On the median FRID map, up to 6 departures may fall within the maximum historical FRI range. See Appendix C for community-specific ranges.

[8] It is important to note that the historical FRI values applied to the Mojave grasslands come from estimates of Native American burning in valley grassland found in the Central Valley and along the southern coast of California. Thus, these estimates may not be appropriate for the desert grassland systems. These FRI values were used to evaluate departures from a hypothetical Native American burning scenario within the Mojave grasslands, and interpretation of these results must consider this uncertainty.

nor the impact of fire on California grasslands is well understood, though it is likely that fires occurred often (Keeley et al., 2011; Wills, 2006). A median FRI of 3 years and a high FRI of 8 years have been estimated for California grassland steppe based on Native American burning practices (Stephens et al., 2007). FRID analysis suggests that the Ranch's grasslands have departed significantly from this estimated fire regime, though the actual burning frequency due to Native American ignitions on the Ranch is not known. The invasion of nonnative grasses in the Central Valley may have caused grasslands to become more fire-prone (Wills, 2006). Valley fire regimes today have been characterized as consisting of surface fires of low

Implications for Wildlife: San Joaquin Kit Fox & Kangaroo Rats

The San Joaquin kit fox (*Vulpes macrotis mutica*) is a federally-listed endangered species that thrives in areas with sparse to moderate vegetation cover (California Wildlife Habitat Relationships System, 2008; Brown et al., 2006). Warrick and Cypher (1998) found that kit fox detections increased after fire, possibly due to decreased habitat suitability for kit fox predators, as well as improved ability of kit foxes to detect predators. Prey availability was not found to play a role in kit fox distribution.

At Tejon Ranch, the kit fox has been found in the Old Headquarters, Comanche Point, and White Wolf areas. These locations are marked by high numbers of kangaroo rats (Cypher et al., 2010), which kit foxes have been observed to feed to their young (Morrell, 1972). Potter et al. (2010) and Price et al. (1995) found higher numbers of Heermann's and Stephens' kangaroo rats, respectively, after prescribed burns, and the BLM recently used controlled burns in the Lake Mathews-Estelle Mountain SKR Reserve grasslands for Stephens' kangaroo rat habitat restoration (Palm Springs South Coast Fuels Program, 2011). Thus, fire in grasslands may benefit kit foxes and kangaroo rats.

intensity and moderate to high severity (Wills, 2006), with a fire season generally occurring between May and October (Reiner, 2007). Under multiple climate models, California's grasslands are predicted to expand, partly as a result of the increased fire spread they encourage (Lenihan et al., 2008). Larger areas may be impacted by grassland fires (Lenihan et al., 2008; Fried et al., 2004), which may be more frequent and less controllable (Fried et al., 2004).

Fire affects nutrient levels and community composition within grasslands.[9] Although the natural breakdown of organic matter occurs slowly in grasslands (Boerner, 1982), fire consumes thatch and makes resources available to growing plants (Vogl, 1979). Some studies have found that bunchgrass species, such as *Nassella pulchra*, are promoted by burning (Dyer, 2002, 2003; Wills, 2006), while other studies have found that *N. pulchra* can be negatively impacted or unaffected by burning (Reiner, 2007; Marty et al., 2005; D'Antonio et al., 2002). Forbs, both native and nonnative, typically increase after fire (Harrison et al., 2003; Keeley et al., 2011; D'Antonio et al., 2002; Pollak & Kan, 1998; Hastings & DiTomaso, 1996; Meyer & Schiffman, 1999; Parsons & Stohlgren, 1989; Dickens et al., 2008; Wills, 2006; Reiner, 2007; Gillespie & Allen, 2004; Hervey, 1949). Species richness may also be augmented by prescribed burning; in areas where nonnative species are dominant,

[9] Research on alkali meadows—grasslands characterized by shallow water tables and relatively high-pH soils—suggests that native grasses may benefit from fire under certain conditions (Hansen, 1986; Pritchett & Manning, 2009). Prescribed burning is actively used in alkali systems for invasive species management (U.S. Fish and Wildlife Service, 2006; Harvey, 2003; Racher & Britton, 2003).

burning may have a greater impact on nonnative richness than native richness (Harrison et al., 2003).

2. Riparian Areas

Riparian communities, including valley and foothill riparian areas, montane riparian forest, sycamore alluvial areas, and desert washes, occur throughout the Ranch (Appelbaum et al., 2010). These productive communities create three-dimensional zones of interaction between streams and upland vegetation (Gregory et al., 1991). With relatively high fuel moisture, humidity, and soil moisture, riparian areas generally have longer FRIs and less intense fires than upland areas (Pettit & Naiman,

Implications for Wildlife: Macroinvertebrates

Macrobenthic communities provide an important food source for organisms higher on the food chain, and influence decomposition, primary production, and nutrient cycling (Wallace & Webster, 1996). Through its impact on macroinvertebrates, fire can affect the functioning of these processes. Streams will usually stabilize 7-10 years after a fire, but shifts in the macroinvertebrate community may persist for longer (Minshall, 2003). These shifts are often due to indirect effects of fires, including effects on light, temperature and food supply. Species composition may shift toward disturbance-adapted organisms and generalists, such as *Baetis bicaudatus* (a mayfly) and *Zapada columbiana* (a stonefly). Species that need specific water quality and flow speed are likely to decline. The result of these changes is often an increase in abundance accompanied by a decrease in diversity (Neary et al., 2005).

2007). Occasional fire in riparian vegetation contributes to long-term nutrient cycling, creates a mosaic of age and vegetation composition, increases sedimentation, and opens up the tree canopy (Neary et al., 2005; Minshall, 2003).

Riparian fire regimes may be changing due to human activity, climate and other influences. On the Ranch, the diversion of water for livestock and agriculture is thought to have pushed riparian fire regimes closer to those of surrounding areas. Invasive plants such as tamarisk (*Tamarix spp.*) can increase fire frequency and severity. In extended droughts, which may become more common as climate change continues, lowered fuel moisture levels and dense vegetation can make riparian areas more vulnerable to high intensity fires, and could even transform them into corridors that drive fire across the landscape (Appelbaum et al., 2010; Pettit & Naiman, 2007; Dwire & Kauffman, 2003).

3. Oak Woodlands

Woodlands dominated by valley oak (*Quercus lobata*), blue oak (*Quercus douglasii*), black oak (*Quercus kelloggii*), canyon live oak (*Quercus chrysolepis*), and other oak species occupy approximately 82,000 acres of Tejon Ranch. Valley oak woodlands cover roughly 7% of the Ranch, and are most abundant at elevations from 400 to 600 meters and from 1400 to 1800 meters (Hoagland et al., 2011). Historically, these areas are thought to have experienced frequent low intensity fires that typically did not kill mature valley oaks. Seedlings and saplings are often top-killed by these low intensity fires, but readily resprout from the root crown. In many oak woodlands, fire suppression has significantly extended FRIs, leading to a buildup of understory fuels and increased risk of destructive high intensity fires. Fire suppression can also promote displacement of valley oaks by live oaks, shrubs, and conifers, although this typically occurs in wetter regions and there is no evidence that it is a problem at Tejon Ranch (Howard 1992a; McCreary, 2004; Standiford & Adams, 1996; Griffin, 1976; F. Davis, personal communication, February 24, 2012). Valley oak woodlands on the Ranch are an average of 7.5 FRIs away from estimated presettlement fire regimes.

Blue oak woodlands cover approximately 6% of the Ranch, with greatest abundance between 500 and 1000 meters in elevation (Hoagland et al., 2011). Like valley oak woodlands, blue oak woodlands are thought to have experienced frequent, low intensity fires prior to European settlement. A study in the Tehachapi Mountains estimated mean FRIs in blue oak woodlands to be 9.6-13.6 years (pre-1843), 3.3-5.8 years (1843-1865), and 13.5-20.3 years (post-1865) (Skinner & Chang, 1996). Blue oak woodlands on Tejon Ranch are an average of 7.4 FRIs away from estimated presettlement fire regimes. Blue oaks are fire tolerant, but wildfire does not appear to be necessary or beneficial for blue oak establishment, growth, or survival, and frequent fire may in fact suppress recruitment (Tyler et al., 2006; Swiecki & Berndardt, 2002).

Black oak woodlands cover approximately 2% of the Ranch, and are found predominantly at elevations above 1200 meters. Black oak woodlands historically experienced a low severity or mixed severity fire regime. Surface fires occurred frequently in the summer and fall, while moderate to high intensity fires occurred less frequently and may have resulted in occasional stand replacement (Fryer, 2007; Keeley, 2006b; Van Wagtendonk & Fites-Kaufman, 2006; Kauffman & Martin, 1986). Black oak woodlands on the Ranch are an average of 10.9 FRIs away from estimated presettlement fire regimes. Black oaks have a number of adaptations to periodic fire, including thick bark, a large root system with ample nutrient reserves, and the ability to resprout from the root crown (Fryer, 2007; Tappeiner & McDonald, 1980). In mixed conifer woodlands where black oak is a co-dominant, shade tolerant conifers can outcompete black oaks during long fire-free periods, eventually excluding them from a site (Fryer, 2007; Swiecki & Bernhardt, 2002; Kauffman & Martin, 1986).

Tejon's canyon live oaks are primarily found in dense woodlands on shady north-facing slopes, but they also dominate south-facing slopes along the upper portions of the Ranch's Blue Ridge (M. White, personal communication, December 20, 2011). Historically, fires occurred in canyon live oak woodlands with an estimated frequency of less than 35 years (Tollefson, 2008; Arno, 2000; Paysen et al., 2000; Skinner & Chang, 1996). Fires occurred primarily in the summer and fall, tended to be of low or moderate severity, and were less frequent in areas of steep terrain (Tollefson, 2008). Canyon live oak woodlands on the Ranch are an average of 4.2 FRIs away from estimated presettlement fire regimes. Canyon live oaks are sensitive to fire—even mature individuals may be top-killed by low intensity fires—but they will readily resprout (Tollefson, 2008; Skinner et al., 2006; Plumb & Gomez, 1983; Minnich, 1980).

4. Montane Conifer Forests

At higher elevations, the Ranch hosts mixed conifer forests and white fir stands. Before European settlement, conifer forests in California were subject to a mixed severity fire regime, with ignitions caused by both lightning strikes and Native American fire management. Low severity ground fires frequently burned through the understory, thinning smaller and less fire resistant saplings. This process maintained tree densities and fuel loads at low levels, and favored retention of mature, fire resistant trees, creating the "park-like" appearance historically associated with these forests (Belsky & Blumenthal, 1997). Localized high severity crown fires also occurred,

Implications for Wildlife: California Spotted Owl

The California Spotted Owl (*Strix occidentalis occidentalis*) has been designated a California Bird Species of Special Concern due in part to habitat loss caused by catastrophic wildfires. The large, mature conifers in which spotted owls nest are resistant to ground fires, but can be destroyed by high severity crown fires (Bond, 2002). This can result in both direct and indirect owl mortality. Low intensity ground fires, on the other hand, are believed to improve owl habitat and increase the abundance of spotted owl prey populations by promoting snags, shrubs, and herbaceous cover (Bond, 2009). Moreover, the absence of ground fire can lead to dense under-stories of shade tolerant species that impede owl foraging and degrade overall habitat quality (Verner et al., 1992). Thus, protecting California spotted owl habitat is likely to entail 1) preserving large, mature trees, 2) retaining large snags and downed woody material for habitat viability, and 3) reducing the risk of stand-replacing fires (Verner et al., 1992).

helping to create the mosaic of age classes associated with healthy forests. These fires exposed mineral-laden soil and created sunlit gaps, fostering the regeneration of shade intolerant conifer species (Habeck, 1992).

The fire regime in montane conifer forests has changed significantly since European settlement. A variety of factors, including grazing, logging and, in particular, aggressive fire suppression, have led to increased fuel loads and forest densities (Van Wagtendonk & Fites-Kaufman, 2006). Our FRID analysis suggests that more than half of the Ranch's conifer stands have departed by at least 10 FRI's from presettlement intervals. Unchecked by fire, shade tolerant species such as white fir and incense cedar become increasingly dense, amplifying severe wildfire risk, aggravating the impact of droughts, and making the forest more susceptible to insect infestation and disease (Zouhar, 2001). Fire suppression policies aimed at preserving forests may have contributed to destructive wildfires: by increasing FRIs, these policies shifted burn patterns from low intensity ground fires to massive, stand-replacing crown fires. Left unchecked, this new regime could potentially shift species composition toward a montane chaparral environment (Wagtendonk & Fites-Kaufman, 2006).

As in other conifer forests, white fir stands were historically subject to mixed severity fire regimes where ground fires burned through the understory and occasional high severity fires created gaps in the forest. Because it is shade tolerant, white fir can readily germinate and grow on the forest floor, and much of the literature treats it as the culprit of stand densification and increased fire risk (Zouhar, 2001). But it also grows naturally in white fir-dominated stands, which provide habitat for species such as the California spotted owl (Verner et al., 1992).

While not as fire tolerant as yellow pines, mature white fir develops thick bark that can resist low intensity ground fires. A tendency to retain lower branches, however, increases the risk of crown fires. Shallow root systems also make white fir susceptible to smoldering ground fires (Mutch & Parsons, 1998). Fire suppression in much of California has increased FRIs and enabled white firs to boost their density, as new recruits are not cleared by ground fires. In addition to increasing fire risk, higher densities reduce the overall health of the forest. The influx of young trees amplifies competition for water, causes drought stress, and can increase vulnerability to disease and pest invasion (Zouhar, 2001). Following severe fire, white fir can regenerate under the shade of shrubs or chaparral species, but full recovery can take many decades (Keeley, 2006b).

5. Chaparral

Chaparral is distributed across a range of elevations along the southeastern side of the Tehachapi Mountain Range and the southern edge of the Ranch property. Stands cover approximately 16,200 acres of the Ranch, roughly 12,900 of which have burned at least once since 1878. Chaparral typically benefits from stand-replacing crown fires at a frequency of approximately 20 to 100 years (Conard & Weise, 1998). Some species respond to fire by producing vegetative sprouts from underground roots or burls. Others germinate from underground seedbanks, responding to cues from heat, smoke, and charred wood (Keeley, 2006b). In southern California chaparral, fires tend to burn during hot summer conditions, or in the fall under extreme fire weather characterized by foehn winds (Stephens & Sugihara, 2006).

Although many chaparral species require fire for germination, chaparral does not significantly degrade in the long term absence of fire. Century old chaparral has been observed to remain as vigorous as and no less diverse than younger stands (Keeley, 1992a). Long intervals between fires may only become a threat if they exceed species longevity, which is more than 100 to 200 years for most chaparral species (Zedler, 1995). Some obligate seeding species, including buckbrush (*Ceanothus cuneatus*) and Mojave ceanothus (*Ceanothus greggii),* may be at risk in the prolonged absence of fire as they succumb to competition from sprouting species, though this risk is not significant up to FRIs of approximately 100 years (Zedler, 1995; Keeley 1992a). Following the occurrence of fire, older stands can provide a

Brewer's Oak

Stands of Brewer's oak (*Quercus garryana* var. *breweri*) occur on the eastern side of the Ranch at an elevation of approximately 5,000 feet. They cover approximately 2,700 acres, over half of which were burned in 1992 during a prescribed fire treatment. Brewer's oak reproduces from both seeds and sprouts to form nearly pure single-species dominated stands. There is limited research on the response of Brewer's oak to fire, but at the species level, mature trees are typically fire resistant and can recover from fire damage by vegetative sprouting. Long fire intervals may lead to encroachment from overtopping conifers, although this has mostly been observed in less arid regions of the Pacific Northwest. Repeated high frequency fires could prevent canopy closure, which may provide a competitive advantage to invasive annual grasses (Gucker, 2007).

competitive advantage to the establishment of obligate seeding species. The accumulation of dead brush during long fire-free periods can result in high intensity fires, causing greater mortality of sprouting species and providing more openings for seedling establishment (Keeley & Zedler, 1978).

At the other extreme, native chaparral vegetation may be lost if fires occur at less than 10 to 20 year intervals (Keeley et al., 2009). High frequency fires can prevent canopy closure and lead to encroachment by invasive annual grasses. Obligate seeding species, which rely on fire-stimulated seed germination, are at the greatest risk if fire frequency exceeds natural FRIs. If repeat fires occur before obligate seeding species mature and produce a viable seedbank, species may not be able to reestablish (Zedler, 1995). Sprouting species are less susceptible to short and long FRIs because they can sprout continuously both in response to and in the absence of fire (Keeley 1992b). They have a competitive advantage over obligate seeding species following the occurrence of fire in young stands. Because there are fewer dead shrubs in young stands, fire intensity is usually less severe, there is less fire-caused mortality of sprouting shrubs, and fewer openings are created for obligate seeding species (Keeley & Zedler, 1978).

A range of fire frequencies can thus promote a diversity of both sprouting and seeding chaparral species (Quinn & Keeley, 2006; Keeley, 1992a; Keeley & Zedler, 1978). An examination of the fire record indicates that time since fire, and consequently the age of chaparral vegetation, is variable in the Ranch's chaparral (see Figure 9). Using time since fire as a proxy for stand age, approximately 25% of stands are younger than 20 years old, approximately 47% are between 21 and 40 years old, and approximately 20% are older than 100 years. This suggests that there may be a diversity of species present, including species promoted by both low and high fire frequencies. However, field surveys are necessary to determine species composition, as time since fire is only one among many factors influencing shrub species establishment.

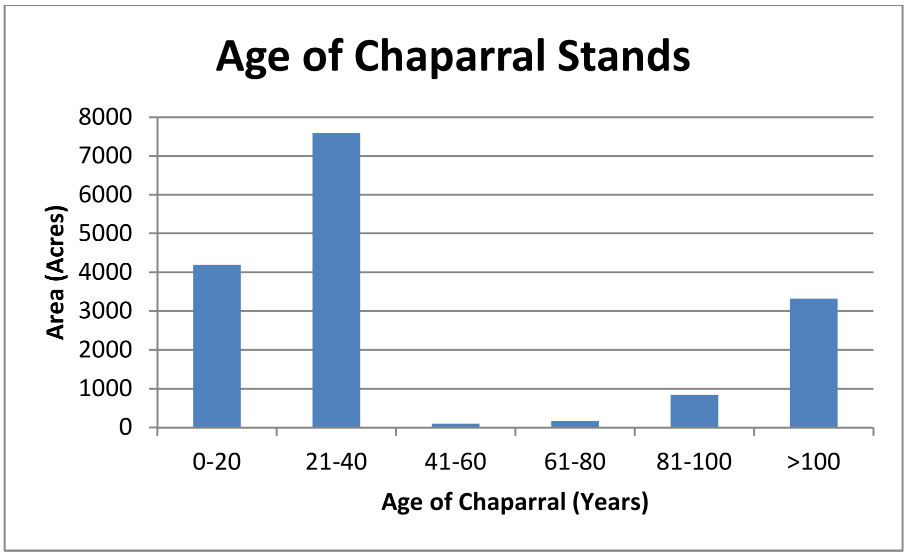


Figure 9: Age of chaparral stands (excluding Brewer's oak) on the Ranch, measured as time since last fire. Approximately 4,000 acres of chaparral burned in prescribed fire treatments in 1987, 1988, 1989 (21 to 40 year age class) and 160 acres in 1992 (0 to 20 year age class).

6. Joshua Tree Woodlands and Desert Scrub

Along its southeastern edge, the Ranch hosts approximately 2,000 acres of Joshua tree woodlands and over 16,700 acres of Mojavean scrub. These communities are dominated by widely-spaced Joshua trees (*Yucca brevifolia*), along with desert shrubs such as beavertail (*Opuntia basilaris*), creosote bush (*Larrea tridentata*) and desert almond (*Prunus fasciculata*) (David Magney Environmental Consulting, 2010). Historically, surface fuel loads have been low, and fire spread has been limited by a lack of horizontal fuel continuity (DeFalco et al., 2009; Brooks & Matchett, 2006; Brooks & Minnich, 2006). The majority of the Ranch's Joshua tree woodlands and desert scrub areas have no recorded fire history since 1878, and presettlement FRIs have been estimated to range from 610 to 1,440 years (Safford et al., 2011). This may be an overestimate for the western Mojave, where intervals appear to have been short enough to select for a strain of Joshua tree (*Yucca brevifolia herbertii*) capable of resprouting after fire (Barbour et al., 2007). But it is not clear that *Yucca brevifolia herbertii* actually benefits from fire and, given the underlying fuel dynamics, fires are unlikely to have been frequent or severe.

Saltbush Scrub

In addition to Mojavean scrub, the Ranch hosts a number of small (<500 acres) saltbush scrub communities, many of which burned with the surrounding grasslands in the Comanche Fire of 2011. These areas are dominated by Atriplex species, including *Atriplex lentiformis* (big saltbush), *Atriplex polycarpa* (common saltbush), and *Atriplex spinifera* (spiny saltbush). Prior to the 1980s, fire management was of little concern in these systems due to the relatively slow-burning nature of saltbush shrubs and lack of continuous fuels (Paysen et al., 2000; West, 1994). Under natural conditions, fire is believed to occur every 35 to 100 years, although the profusion of invasive annual grasses may increase fire risk (Meyer, 2005; Paysen et al., 2000). The response of saltbush scrub species to fire is poorly documented, suggesting that periodic surveys of recently-burned saltbush scrub could provide useful insights into postfire species establishment and community structure.

This fire regime may be altered by the spread of disturbance-exploiting invasive annual grasses. Elsewhere in the Mojave, species such as cheatgrass (*Bromus tectorum*) are creating a continuous layer of fine fuel cover, thereby facilitating fire spread. Because these species regenerate quickly, and are adapted to a high frequency, high severity fire regime, their spread can lead to a self-perpetuating cycle of increased fire, followed by more abundant invasives. In parts of the Mojave, this cycle has advanced so far that invasive annual grasses form a majority of plant biomass[10] (Brooks et al., 2011; Cole et al., 2011; DeFalco et al., 2009; Brooks & Matchett, 2006; Brooks, 2000). The Ranch's deserts are relatively far from this outcome (and its Joshua tree woodlands appear to be expanding), but invasive grasses have clearly established a foothold. If future disturbances provide opportunities for them to expand, the invasive-wildfire cycle is likely to progress further (Appelbaum et al., 2010).

V. MANAGEMENT APPROACHES AND RECOMMENDATIONS

A. MANAGEMENT APPROACHES CONSIDERED

1. Grazing

As a fire management tool, grazing has complex long-term effects on a number of vegetation communities (Stahlheber & D'Antonio, 2011; D'Antonio et al., 2002; Germano et al., 2001). Its immediate effect, however, is generally to reduce

[10] This cycle also affects desert wildlife. Increased wildfire frequency, for example, has been identified as a threat to desert tortoise (*Gopherus agassizii*) habitat and food supply (U.S. Fish and Wildlife Service, 2011b; Esque et al., 2003). It has also been linked to decades-long reductions in biodiversity among small mammals in Joshua tree woodlands (Vamstad & Rotenberry, 2009).

the abundance and continuity of surface-level fine fuels (Leonard et al., 2010; Holdo et al., 2007). By removing fine fuels, it reduces the overall amount of fuel, while increasing the packing ratio of what remains (Husari et al., 2006; Van Wagtendonk, 2006). In grassland communities where most above-ground biomass consists of palatable fine fuels, grazing can reduce the frequency and severity of wildfires, slow their spread, and ultimately shorten the fire season (Leonard et al., 2010; Holdo et al., 2007; Huntsinger et al., 2007; Husari et al., 2006). The amount of residual dry matter (RDM) remaining at the end of the season impacts shade and soil conditions, and thereby affects the next-year's germination (Huntsinger et al., 2007). In savannahs and forests, grazing can reduce horizontal, and sometimes vertical, fuel continuity. It can also interact with fire on smaller spatial scales, as when intense grazing is used to create firebreaks at wildland-urban interfaces (Husari et al., 2006).

The effects of grazing on fuel levels depend on a number of factors, but Huntsinger et al. (2007) posit that only four are within managers' control: 1) animal type (including not only species, but also age and physical condition); 2) the distribution of grazers across the landscape; 3) the timing and duration of grazing; and 4) the density of grazers. These factors can be manipulated individually or in combination to control fuel levels. For example, managers can select grazers that prefer particular fuel species, or time grazing to interact with the phenology of these species (Huntsinger et al., 2007).

2. Mechanical Thinning

Manual and mechanical fuel treatments can mimic certain fire effects by reducing, rearranging, or otherwise modifying fuel loads. Fuel reductions have shown the greatest value in forests with low and mixed severity fire regimes (Keeley et al., 2009). Manipulation of fuels to achieve a desired vegetation structure is not as useful in crown fire systems, such as chaparral, where fuel accumulation is not the cause of large fires (Agee & Skinner, 2005). Mechanical treatments are often used as a precursor to prescribed fire; the removal of biomass can create conditions that are favorable for a controlled burn (Graham et al., 1999). Enhanced precision and control, minimal impact on air quality, and broader social acceptance may make mechanical treatments preferable to prescribed fire under certain conditions (Husari et al., 2006; Hoshovsk & Randall, n.d.).

Mechanical treatments are an imperfect surrogate for fire, however, and will not produce all of the same ecosystem benefits. Moreover, treatments are limited by steep slopes and inaccessible locations, can severely damage soils and vegetation, and promote invasive species (Keeley et al., 2009; Husari et al., 2006; Hoshovsk & Randall, n.d.). The ultimate effectiveness of mechanical fuel manipulations depends in part on how the residual fuels are treated (Keeley et al., 2009). If residual fuels are not treated or removed, overall fire hazard may not be reduced. Treatments such as mastication, piling, and surface prescription burns can reduce hazards related to residual surface fuels (Bartuszevige & Kennedy, 2009; North et al., 2007).

Because it can reduce tree density and eliminate ladder fuels, thinning may be particularly useful in conifer forests. Removing smaller trees and saplings can help restore the "park-like" conditions thought to have existed in conifer forests before European settlement. Thinning can also build resilience to drought and disease, as trees in less dense forests face reduced competition for water (Ma et al., 2010). A significant drawback, however, is that an estimated 20-50% of remaining trees can be wounded during a mechanical thinning operation (Zouhar, 2001).

3. Prescribed Fire and Wildland Fire Use

Prescribed fire is used to achieve a number of management goals, and was applied on the Ranch in 1987, 1988, 1989 and 1992. It can reduce fuel loads to prevent high severity fires (e.g., van Mantgem et al., 2011; Potts et al., 2010; Fry, 2008; Moghaddas et al., 2008), release nutrients, provide diverse habitat for wildlife (Biswell, 1989), create forest canopy gaps (Schmidt et al., 2006), and promote fire-dependent plant species (Husari et al., 2006). But prescribed fire has numerous drawbacks, including a potential influx of nonnative species (e.g. Keeley, 2006a; Knapp et al., 2007), impaired air quality (California Environmental Protection Agency, 2003), and risk of escape (e.g. National Park Service, 2011). Fire in general has also been associated with increased erosion (Wohlgemuth et al., 2006), and may adversely affect certain wildlife (e.g. Dwire et al., 2011).

Wildland fire use, the practice of permitting a fire caused by lightening to burn, is used to allow fire to function as a natural component of ecosystems (van Wagtendonk, 2007; Miller, 2003). This practice can reproduce elements of natural fire regimes, alter forest conditions and reduce fuel loads, such that subsequent wildfires are limited (Collins & Stephens, 2007; Collins et al., 2009; Fule & Laughlin, 2007; Laughlin et al., 2004; Miller, 2003).

Potential drawbacks include the loss of control over fires (Joint Fire Science Program, 2009), the influx of invasive species such as yellow star thistle, air pollution, and damage to sensitive species. In addition, prior fuel removal may be required in some cases (Miller, 2003). It has been suggested that wildland fire use is most applicable in large, isolated areas where risks to developments or infrastructure are minimal (Joint Fire Science Program, 2009). Many of the Ranch's vegetation communities are patchy and limited in size, making it difficult to allow a fire to burn with limited control in some areas while excluding it from more fire-sensitive ecosystems. Moreover, on a working ranch with a number of developed areas, wildland fire use may be difficult to implement safely.

4. Herbicide Use

Chemical control of vegetation with herbicides can inhibit the growth or establishment of undesired, fire regime-altering species. Herbicides are commonly applied to undesired species that would otherwise resprout vigorously following a prescribed burn or mechanical treatment. While herbicides pose some environmental risks, these risks can be minimized by selecting compounds which degrade rapidly, are not poisonous to animals, do not easily volatize, and are immobilized by soil particles (Hoshovsk & Randall, n.d.). Broadcast treatments with selective herbicides may pose a greater environmental risk than direct application methods, but can be more economical over a large scale (Kyser et al., n.d.; DiTomaso et al., 2007). The effectiveness of herbicide treatments can vary considerably with geographic and environmental conditions, as well as varying levels of tolerance among populations of the same species. Given this inherent variability and limited data documenting the effects of herbicides in wildland areas, trials should be conducted over small plots before herbicides are used in wildland vegetation management (Wigley et al., 2000; Hoshovsk & Randall, n.d.).

5. Revegetation

Revegetation with desired plant species can help counter the reestablishment of undesired, fire regime-altering plants following their removal in treatment areas. Whether revegetation can achieve a desired species composition or structure depends on a number of factors. Seeded and planted species must be well-adapted to conditions at the treatment site (DiTomaso et al., 2007). For the greatest chance of successful establishment, seeds or cuttings should typically originate from the treatment site or an adjacent area (although this is complicated by climate change) (Hoshovsk & Randall, n.d.). Risks associated with seeds and plants collected from distant locations include project failure, introduction of diseases, and loss of genetic diversity (Hoshovsk & Randall, n.d.). Regardless of their origin, species that establish successfully and outcompete undesired vegetation in one treatment area may fail to establish under different geographic, climatic, and environmental conditions.

The practice of broadcast seeding in order to rehabilitate a site after the occurrence of fire is generally discouraged (Thode et al., 2006). Native plant communities will often reestablish without management intervention following fires, and postfire broadcast seeding can promote invasive species establishment (Erickson et al., 2007; Keeley, 2006a).

B. Modeling of Management Scenarios Using LANDIS-II

Computer modeling can be a useful tool to explore the potential effects of alternative fire management scenarios and to gain insight into how different variables might affect fire regimes. We used LANDIS-II, a stochastic, spatially-explicit forest succession and disturbance model, to simulate the potential effects of residential development, climate change, and fire management across a portion of Tejon Ranch. We developed and modeled scenarios to examine three different questions:

1) How might thinning and prescribed burning affect fire size, frequency, and severity in conifer forests?
2) How might climate change affect fire size, frequency, and severity?
3) How might the construction of Tejon Mountain Village affect fire size, frequency, and severity?

1. Model Background

LANDIS-II simulates ecological succession and disturbance, and has been successfully used throughout the United States to simulate fire and other forest disturbances (Scheller et al., 2011; Sturtevant et al., 2009; Scheller et al., 2008). It is designed to simulate succession over long time scales on landscapes larger than 10,000 hectares (Scheller et al., 2007). It differs from other forest models in that it simulates age cohorts of each tree species, rather than individual trees, and allows the user to set variable time steps for each forest process. The base model uses life history attributes of dominant plant species to model forest succession. In addition, users can select from a variety of optional extensions which model specific processes, such as fire, in greater detail.

2. *Summary of Methods*[11]

a. *Extensions Used*

In addition to the base model, we used the Dynamic Fire System, Dynamic Fuel System, Biomass Succession, and Biomass Harvest extensions. The Dynamic Fire System extension was used to simulate fire across the landscape using weather, fuel, and topographical inputs. The Dynamic Fuel System and Biomass Succession extensions were used to incorporate more complex vegetation dynamics into the fire model. The Biomass Harvest extension was used to simulate different management treatments such as thinning and prescribed burns.

b. *Identifying and Calculating Inputs*

LANDIS-II requires a multitude of user-defined inputs for each extension. We obtained many of the inputs from scientific literature or data specific to Tejon Ranch. Other inputs were calculated using more complex methods, such as MaxEnt, described in Appendix D.

c. *Region of Interest*

The region of interest (ROI) used in the LANDIS-II simulations was a 32,606-

[11] A detailed description of the methods used is provided in Appendix D.

acre (13,195 ha) polygon on the eastern side of the Ranch (see Figure 10).[12] This ROI was selected because it encompassed most of the conifer forests on the Ranch. Conifer forests were a logical focus for modeling because they are the areas where active fuel management strategies such as prescribed burning or mechanical thinning are expected to have the greatest potential benefit. Potential effects of the Tejon Mountain Village (TMV) residential development, which is planned for an area directly west of the ROI, were explored through an increased ignition probability near the development. A 2 km buffer was placed around the ROI to account for edge bias (Pommerening & Stoyan, 2006).

Figure 10: Region of interest (enlarged on right), showing conifer forests in green.

[12] Because of limitations in computing capacity and data availability, it was not possible to run LANDIS-II simulations for the entire Ranch.

d. Scenarios

We modeled a total of 10 scenarios, summarized in Table 1. Each management scenario was simulated under current climate as well as both climate change scenarios. We modeled a single land use change scenario (under current climate) to simulate the effect of increased ignitions that might occur as a result of the development of Tejon Mountain Village. All scenarios were modeled until 2060.

To model climate change, we used a single emissions scenario, A2, and two Global Circulation Models (GCMs): 1) the National Center for Atmospheric Research's Parallel Climate Model (PCM); and 2) the National Oceanic and Atmospheric Administration's (NOAA's) Geophysical Fluid Dynamic Laboratory (GFDL) model. The PCM predicts that, by the end of the century, the climate in southern California will be warmer and wetter, whereas the GFDL predicts that it will be warmer and drier (Cayan et al., 2008).

We modeled two management alternatives: 1) the thinning of young age cohorts of specific species; and 2) thinning followed by prescribed burning. These alternatives were modeled within selected management areas dominated by conifer species (see Figure 11).

	No Management	Thin Only	Thin and Burn	Land Use Change
GFDL	X	X	X	
PCM	X	X	X	
Current Climate	X	X	X	X

Table 1: Scenarios modeled in LANDIS-II.

Figure 11: Management areas within the region of interest.

3. Results and Analysis

a. Overview

There were no significant differences between any of the scenarios in terms of average fire return interval, average fire size, average number of fires, or average fire severity (see Figure 12). However, visual comparison of the annual probability of burning did reveal potential spatial patterns among the different scenarios (see Figure 13).

Our ability to draw conclusions from the LANDIS-II model was constrained by the number of replicates of each scenario that we were able to run given the available computing capacity. It is possible that we might have discovered statistically-significant differences among the scenarios had we been able to run a large number of replicates of each scenario.

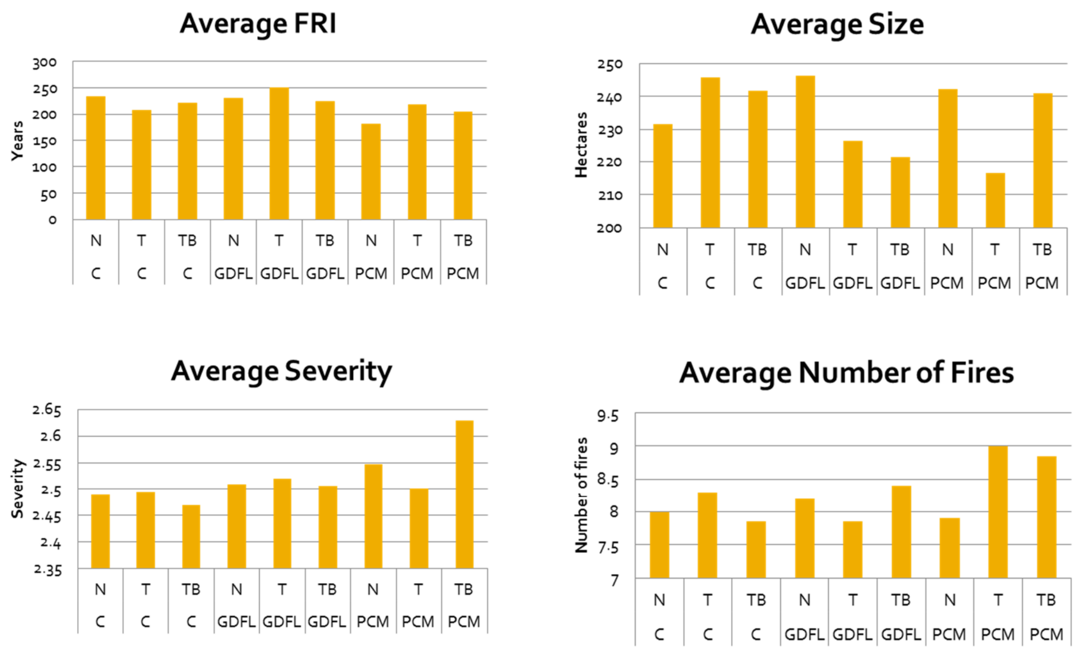
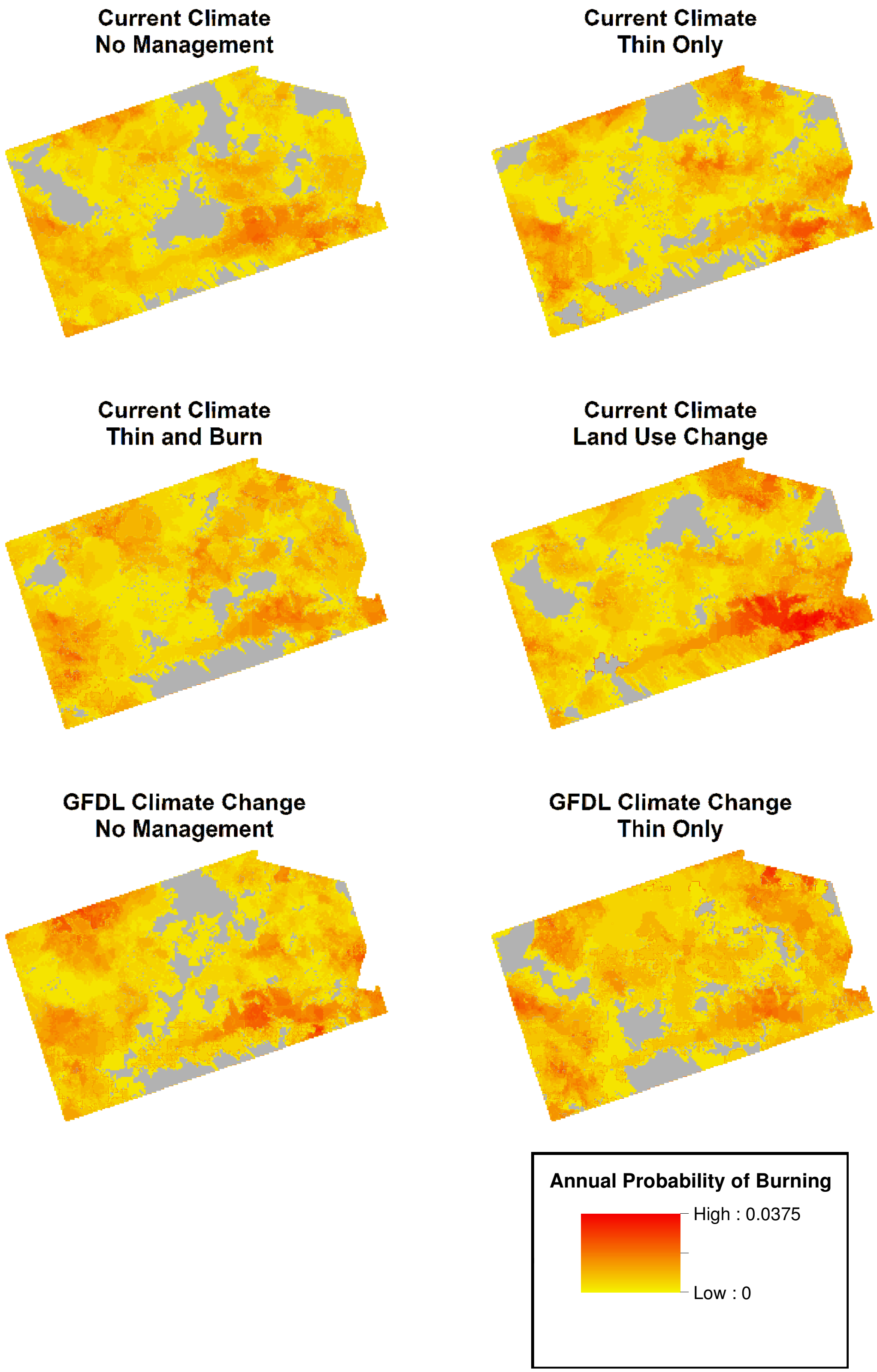


Figure 12: Fire statistics comparing all 10 scenarios. The top letter represents No management (N), Thin only (T), or Thinning followed by prescribed Burns (TB). The bottom letter represents the climate scenario with Current climate (C), the GDFL scenario, and the PCM scenario.


Current Climate
No Management
Current Climate
Thin Only
Current Climate
Thin and Burn
Current Climate
Land Use Change
GFDL Climate Change
No Management
GFDL Climate Change
Thin Only
Annual Probability of Burning
High : 0.0375
Low : 0

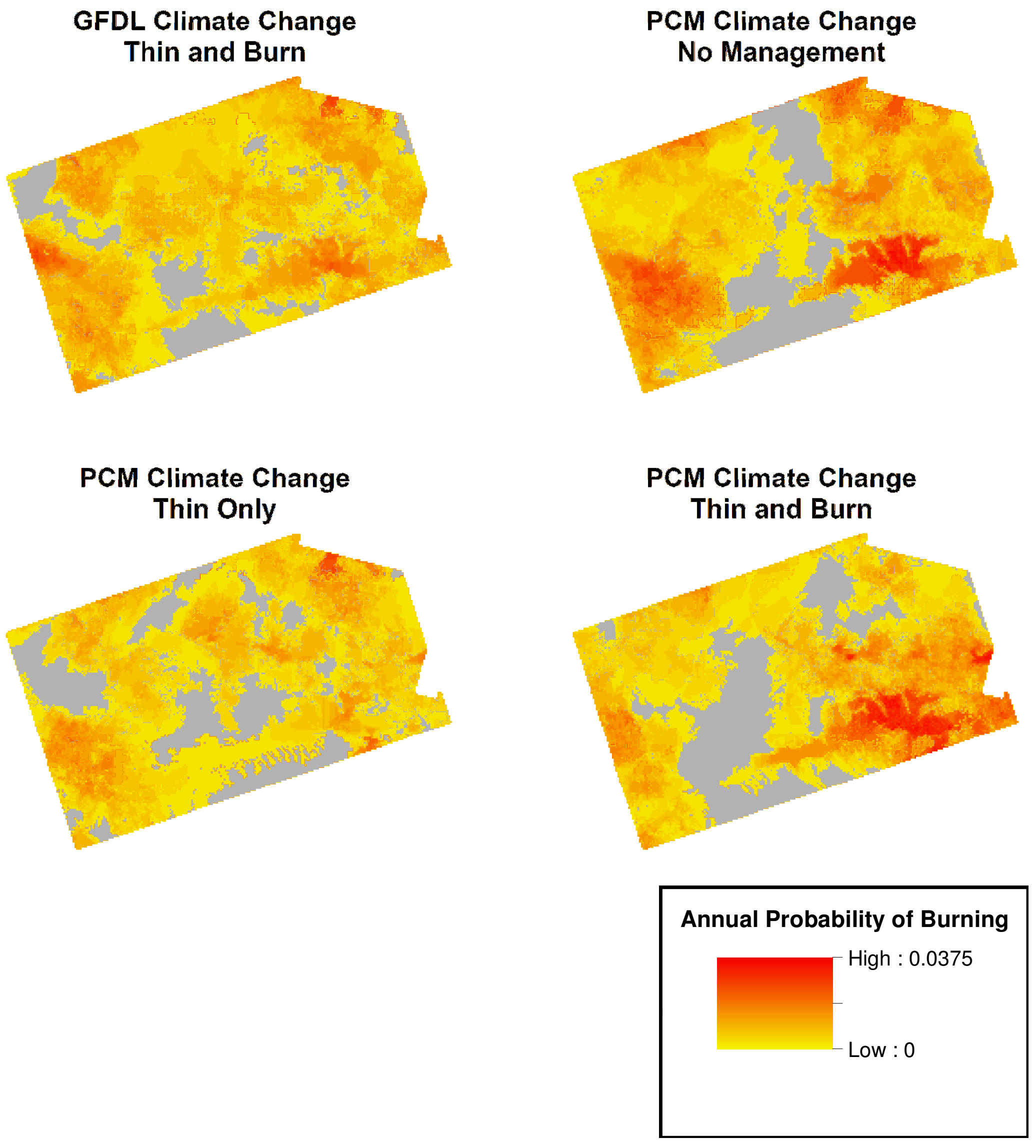


Figure 13: Annual probability of burning over the full 50-year model duration for each scenario, averaged across 10 replicate simulations for each scenario.

b. Effect of management

Neither thinning nor a combination of thinning and prescribed burning were found to affect the annual probability of fire within the management areas (see Figure 14). This result is not necessarily inconsistent with what would be expected from fuel load reduction in conifer forests. Fuel treatments applied in these areas would likely reduce the risk of high-severity, stand-replacing crown fires, but might have a neutral or positive effect on low-severity ground fires, and thus might not reduce the overall frequency of fire. It is possible that the simulated fires that occurred in the management areas were in fact less severe than the fires that occurred outside of these areas, although we were unable to test this hypothesis.

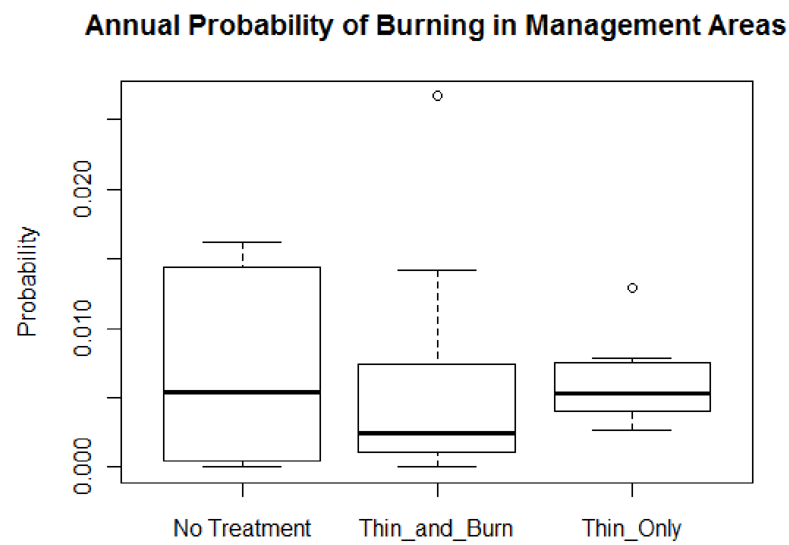


Figure 14: Annual probability of burning in management areas under current climate.

c. Effect of climate change

Our simulations showed small, but statistically insignificant, increases in size, severity, and number of fires for the climate change scenarios compared with the current climate scenario. Visual interpretation of the spatial output also showed an increase in annual probability of fire, especially in areas already prone to fire (see Figure 13). This result is in accordance with the expected effects of climate change in southern California (Lenihan et al., 2008). The maps also show fire to strongly spread along slopes and ridgelines, reinforcing the conclusion that the Ranch's topography strongly influences fire regimes.

Areas dominated by Brewer's oak chaparral exhibited a notable increase in annual probability of fire under the climate change scenarios. Brewer's oak is a valued ecosystem, but little is known about its fire ecology.

d. Effect of land use change

Surprisingly, despite simulating an increase in ignition rates on the western side of the ROI (adjacent to the future Tejon Mountain Village development), the fire frequency in this portion of the ROI did not appear to increase in the land use change scenario relative to the baseline scenario. Because LANDIS-II models fire ignition and fire initiation as separate events, it is possible that an increase in fire ignition rate did not actually lead to an increase in fire initiation rate in this area.

C. COST ANALYSIS

1. Background

Wildfires on Tejon Ranch are currently suppressed by state and county fire departments. While this approach does not create direct costs for the Conservancy, some Ranch ecosystems can be impaired by the absence of fire. Extinguishing fires is expensive, as evidenced by the Comanche Fire of 2011. Both CAL FIRE and the Kern County Fire Department (KCFD) fought this fire; suppression efforts involved more than 600 firefighters and ultimately cost more than $7 million (CAL FIRE, 2011). This recent example, along with numerous studies, suggests that costs associated with fuel management can be reduced in the long term by preventing catastrophic fires (Verner et al., 1992). For this reason, it may be economical for fuel management costs to be shared between the Conservancy and government agencies.

2. Methods

Because conifer systems are the most likely targets of active fuel management, we focused our quantitative analysis on these systems. Our goal was to compare the cost and associated fuel reductions for three different management strategies:

1) Hand thinning of trees less than 10" dbh;
2) Mechanical thinning of trees less than 25" dbh; and
3) Combined treatment (hand thinning followed by prescribed burning).

We began by researching thinning costs in comparable systems around California. Since we suspect that the drier conditions at Tejon have led to less fuel buildup, we based our analysis on costs from the lower end of the range (see Table 2).[13]

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[13] Much of the literature discusses mechanical thinning treatments that involve the harvest and sale of timber and wood products to help pay for the thinning treatment. We chose to disregard this approach due to conservation concerns and the limited size of the Ranch's conifer tracts.

Management Strategy[14]	Cost per Acre	Average Fuel Reduction
Mechanical Thinning <25" DBH (MT)	$1,700	31 tons/acre
Hand Thinning <10" DBH (HT)	$650	9 tons/acre
Combined Hand Thin + Burn (TB)	$950	15 tons/acre

Table 2: Management approaches, costs, and fuel reduction for conifer systems.

Because the primary goal in Tejon's conifer systems is to avoid type conversion associated with crown fire, we used tons of fuel reduced per acre (including both standing and downed fuel) as a metric for evaluating the relative success of management options. The Forest Vegetation Simulator program was used to determine how much fuel is typically removed by these techniques (U.S. Forest Service, 2011). This program allowed us to manipulate treatment parameters to mimic our management scenarios, and then to model these management actions on input data from 88 representative conifer stands throughout California. This provided output that estimated fuel reduction in tons per acre across the modeled stands. Data was tallied and averaged to find the general fuel load reduction associated with each treatment.

We then determined the spatial extent over which these strategies could be applied (see Figure 15). Our analysis focused on the eastern section of the Ranch that encompasses approximately 3,600 acres of conifer forest (the majority of the Ranch's conifer systems). Despite the small economies of scale that accrue as treatment area increases (González-Cabán & McKetta, 1986), treating all of Tejon's conifers would

[14] Sources: National Park Service, 2004; Holl, 2007; U.S. Forest Service, 2011.

likely be cost-prohibitive. Additionally, the Conservancy may initially opt to apply fuel treatments on a smaller scale, so that the impacts of treatment can be studied over time. For these reasons, we used Digital Elevation Models and aspect mapping in GIS to target specific, fire-prone conifer areas with drier and more abundant fuels. This included south facing slopes and ridge tops above 1800 meters, where the Conservancy may choose to focus initial management actions. Analysis of these features indicated that 811 acres were at higher altitude, 204 acres occupied southern slopes, and 38 acres were both on a south facing slope and above 1800m in elevation. To make a more visually useful cost curve, we chose to analyze spatial areas of 38, 90, and 150 acre sections.

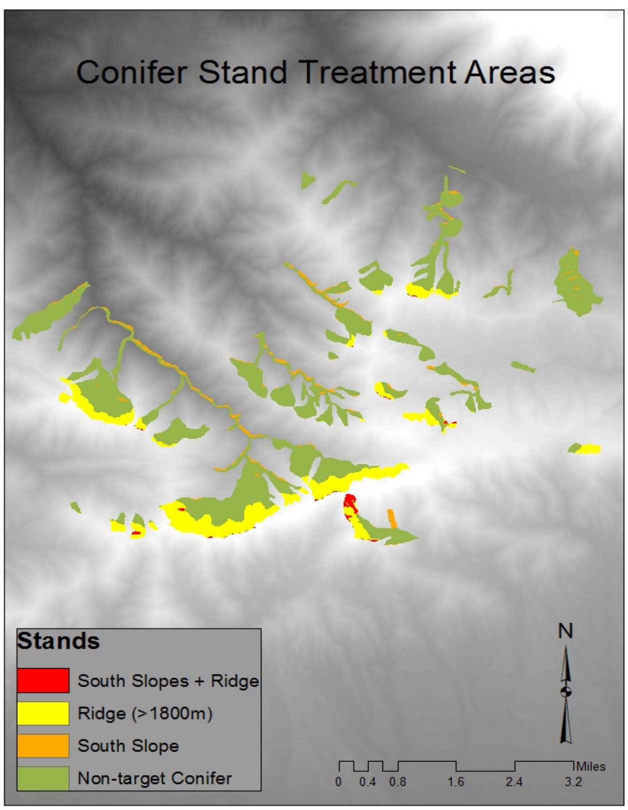

Figure 15: Targeted Conifer Treatment Areas

3. Results and Analysis

With this data, we plotted the cost and associated fuel reduction of the three management strategies, as applied over three spatial extents. Increasing costs were a function of both the type of treatment, and the acreage over which the treatment was applied. Perhaps most importantly for the Conservancy, we found that when less damaging thinning strategies, such as hand-thinning and thinning-plus-burning, are conducted across a larger spatial area, they will provide greater fuel reduction than mechanical thinning in a smaller area (see Figure 16). While extensive hand thinning would be more expensive, it could have less impact on the environment in terms of soil compaction, and cause less collateral damage to desirable mature trees.

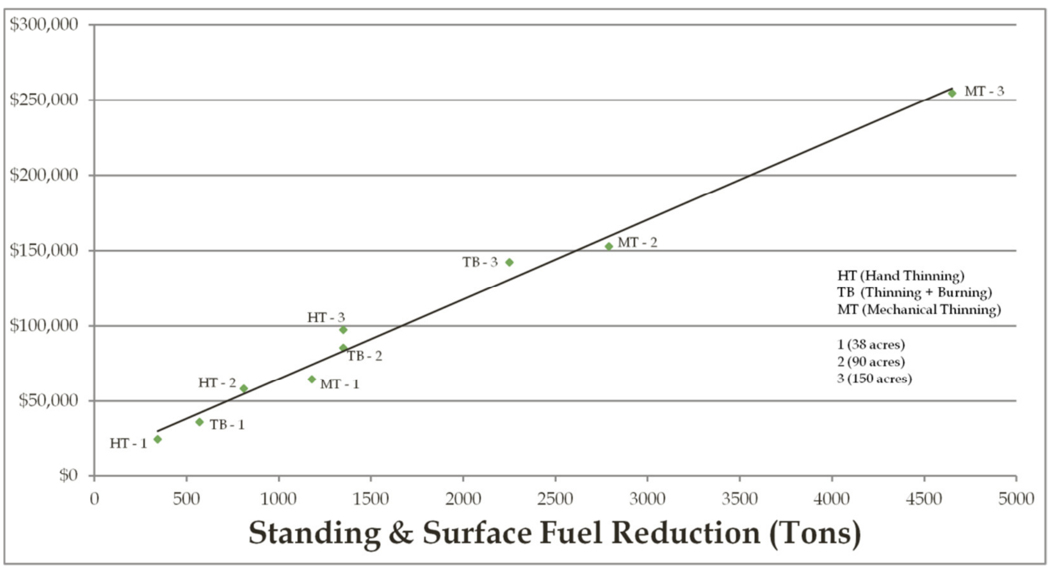

Figure 16: Cost curve for conifer management strategies.

D. MANAGEMENT RECOMMENDATIONS

Results from our research, FRID mapping, LANDIS-II modeling, and cost analysis were integrated to develop management recommendations for each major ecosystem on the Ranch.

1. Grasslands

a. Continue fire suppression.

Sustaining native species populations and maintaining biodiversity in the Ranch's grasslands are the two major targets for grassland fire management. Research has shown that fire can reduce nonnative species dominance in California grasslands under certain circumstances (D'Antonio et al., 2002; Dickens et al., 2008.; DiTomaso et al., 2006; Gillespie & Allen, 2004; Pollak & Kan, 1998; Menke, 1992; Meyer & Schiffman, 1999; Parsons & Stohlgren, 1989; Reiner 2007). Perennial bunchgrasses may respond positively to burning (Dyer, 2002, 2003; Wills, 2006), and native forbs have been observed on many occasions to increase after fire (Harrison et al., 2003; Keeley et al., 2011; D'Antonio et al., 2002; Pollak & Kan, 1998; Hastings & DiTomaso, 1996; Meyer & Schiffman, 1999; Parsons & Stohlgren, 1989; Dickens et al., 2008; Wills, 2006; Reiner, 2007). Fire also facilitates nutrient cycling and can stimulate growth as biomass is removed (Vogl, 1979; Heady, 1956).

But considering the potential drawbacks of burning—including cost, uncertain outcomes, and the persistence of invasive species—prescribed fire is not recommended as a default approach for the Ranch's grasslands. Furthermore, a let-burn policy is constrained by the potential for loss of control. Research has suggested that native species do not always respond positively to fire (D'Antonio et al., 2002; Marty et al., 2005; Reiner, 2007; Wills, 2006), and studies have generally focused solely on *N. pulchra* when considering the effects of fire on native bunchgrasses (e.g., Dyer, 2002, 2003; Gillespie & Allen, 2004; Menke, 1992; Marty et al., 2005). *N.*

pulchra is not one of the species of bunchgrass, such as *N. cernua*, identified in the recent survey of the Ranch's grasslands (Bartolome et al., 2010-2011). Though it is known that excessively frequent fire can deplete nutrient supplies (C. D'Antonio, personal communication, December 8, 2011; Menke, 1992), the impacts of burning over long periods of time have not been evaluated (F. Davis, personal communication, 2012). Furthermore, compositional shifts benefitting native species can be minor (D'Antonio et al., 2002; Keeley, 2001), and nonnative forbs (D'Antonio et al., 2002; Gillespie & Allen, 2004; Harrison et al., 2003; Keeley et al., 2011; Parsons & Stohlgren, 1989; Pollak & Kan, 1998; Wills, 2006; Dickens et al., 2008; Reiner, 2007) and even nonnative annual grasses can be stimulated by fire in some cases (D'Antonio et al., 2002). Annual grasses will also likely recover quickly once burning ceases (D'Antonio et al., 2002; Dickens et al., 2008; Menke, 1992; Meyer & Schiffman, 1999; Parsons and Stohlgren, 1989).

Fire has also been a part of California's desert grassland communities, but research examining its impact on these communities is largely absent. It is possible that grazing and fire exclusion have led to the conversion to nonnative grasslands in some areas of the desert in California (Vogl, 1995). However, there is a significant presence of native species in the Antelope Valley grasslands on the Ranch (Bartolome et al., 2010-2011). The proximity of the Antelope Valley grasslands to fire-sensitive desert systems, as well as the uncertainty surrounding the function of fire in California desert grasslands, precludes the addition of fire to these grasslands.

b. Monitor native species under a grazing regime.

In addition to fire suppression, the predominant force affecting the fire regime on Tejon Ranch is grazing (see Table 3). It is clear that native populations have persisted to some extent since the inception of grazing on the Ranch in 1860, and the Ranch's grasslands may be as "native species-rich" as grasslands found in California state parks (Bartolome et al., 2010-2011). FRID analysis indicates that substantial portions of grasslands in both the San Joaquin and Antelope Valleys have not been experiencing fire on a regular basis, suggesting that grazing and the absence of fire has not led to extirpation of these native populations.

Under the regime of fire suppression and grazing on the Ranch, fine fuel reduction is likely accomplished by cattle, helping to prevent the fast-spreading fires (Huntsinger et al., 2007) that invasive annual grasses may facilitate (Reiner, 2007). Like fire, grazing also influences the assemblage of species that will be present, thereby affecting the fire regime (F. Davis, personal communication, 2012).

Thatch removal accomplished by grazing could support the growth of certain native grassland species (Harrison et al., 2003; Heady, 1956; Menke, 1992). A meta-analysis of grazing in California's grasslands suggests that possible impacts of grazing on Ranch grassland vegetation include increased native forb cover and richness, nonnative grass richness, and nonnative forb cover. Decreased nonnative grass cover could also occur if grazing happens during the wet season (Stahlheber & D'Antonio, 2011). However, in dry environments such as those found on the Ranch, grazing could negatively impact native species, reducing both richness and percent cover (Kimball & Schiffman, 2003;[15] P. Schiffman, personal communication, January 30, 2012). Thus, consideration of the grazing regime that will best support the Ranch's native grassland species is warranted.

We recommend regular monitoring of San Joaquin Valley grasslands to evaluate the stability of native forb and grass populations over time. If decreasing trends are observed under a grazing regime, then long-term test plots in the Ranch's San Joaquin Valley grasslands could be used to assess the outcome of prescribed burns, which could be evaluated with both the presence and absence of grazing (see Appendix A for discussion of the possible combined effects of burning and grazing).

In particular, some geophytes have exhibited sensitivity to grazing, such as reduced frequency (Harrison et al., 2003) or a reduced number of flowering stalks with grazing after fire (Borchert & Tyler, 2009). In one example, Kimball and Schiffman (2003) observed *D. capitatum* only in an ungrazed portion of the Carrizo Plain when comparing a grazed and ungrazed plot. More geophyte species were typically found in grazed areas in a study conducted in Israel, but in a limited number of cases the opposite trend was observed, leading the authors to recommend grazing restriction in some areas to protect those few species impaired by grazing (Noy-Meir and Oron, 2001). Research in Australia has also highlighted a potential negative association between increasing sheep grazing intensity and geophyte occurrence[16] (Dorrough and Scroggie, 2008). If certain geophytes species on the Ranch are

[15] This study was unreplicated.
[16] This result may be somewhat complicated by positive covariation between phosphorus levels and grazing.

thought to be sensitive to grazing, it is recommended that grazing exclosures be tested. Fire could then be considered as an option to prevent fuel accumulation and the proliferation of invasive annuals. Native Americans conducted burning to facilitate geophyte growth and collection and returned to the same collection sites repeatedly, suggesting that these species are adapted to anthropogenic burning (Anderson, 1997). In addition, some geophytes such as *Bloomeria crocea* and *Chlorogalum pomeridianum* are argued to be adapted to fire because they respond to fire in chaparral with extensive flowering, potentially due to increases in light (Tyler & Borchert, 2007; Borchert & Tyler, 2009). However, research of the chaparral geophyte *Zigadenus fremontii* also indicates that the impact of burning may be dependent upon the interval between fires, during which geophytes may replenish carbohydrate stores (Tyler & Borchert, 2002).

San Joaquin and Antelope Valley Grasslands		
Goals	Sustain populations of native grass and forb species. Maintain biodiversity.	
Monitoring Regime/Metrics	• Native species richness and cover • Persistence of focal species populations	
Mgmt Strategy	Grazing	Prescribed Burning
Potential Benefits	Fine fuel reductions. Increased native forb populations. Increased native grass cover and decreased nonnative grass cover with wet-season grazing. Reductions in thatch and invasive seed.	Nutrient cycling and soil enrichment. Increases in native forbs, including geophytes. Positive impacts on bunchgrasses. Reductions in nonnative species.
Cost/acre	Maintenance of grazing infrastructure (presumably more than offset by revenues from grazing leases).	$45/acre. Also, loss of grazing revenue to Ranch if grazing halted.
Where	San Joaquin and Antelope Valley grasslands	San Joaquin Valley grasslands
When/Season	Early spring reduction in both native and nonnative annuals and their seedbanks. Reduction in thatch with spring and summer grazing.	Most support for late spring burning to eliminate nonnative seeds before they are released.
Frequency	Annual.	Intervals of less than 5 years.
Potential Drawbacks	Negative impacts to some geophytes. Suppression of native species in arid environments. Reduction in native forb richness. Smaller bunchgrass (based on *N. pulchra* study) seeds with lower probability of germination.	Loss of control. Negative impacts to air quality. Growth of some nonnative species. Reduction in nutrient stores over time. Potential damage to perennial grasses (based on *N. pulchra* study). Annual grasses may quickly regain dominance if burning is stopped. Unknown impacts of long-term or repeat burning.

Recommendation	Monitoring grazing impacts. Consider excluding where rare geophytes impacted.	Consider test plots if natives declining over time. Consider burning if grazing harmful to rare geophytes.[17]

Table 3: Management strategies for grasslands.

2. Riparian Areas

a. Assess tamarisk stands.

The Ranch contains at least 14 acres of tamarisk, an invasive plant that has been shown to increase fire intensity and frequency (Knapp, 2010; Dwire & Kauffman, 2003; Zavaleta, 2000). Because tamarisk plants can grow up to 30 cm per year, and because the entire population has not been surveyed, the stands on Tejon should be evaluated before they spread further (Knapp, 2010; DiTomaso, 1998). If the assessment finds extirpation to be infeasible, a control plan should be implemented. Control measures can include the prevention of disturbance in nearby areas to minimize spread, as well as continuing monitoring (see Table 4). If extirpation is found to be feasible, monitoring and treatment should continue after the initial removal. Successful removal of tamarisk usually requires a combination of methods, such as mechanical removal combined with herbicides and revegetation.

[17] Sources: Huntsinger et al., 2007; Stahlheber & D'Antonio, 2011; Menke, 1992; D'Antonio et al., 2002; Harrison et al., 2003; Keeley et al., 2011; Kimball & Schiffman, 2003; Borchert & Tyler, 2009; P. Schiffman, personal communication, January 30, 2012; Dyer, 2002; Boerner, 1982; Vogl, 1979; Dyer, 2003; Wills, 2006; Dickens et al., 2008; DiTomaso et al., 2006; Hastings & DiTomaso, 1996; Pollak & Kan, 1998; Gillespie & Allen, 2004; Meyer & Schiffman, 1999; Parsons & Stohlgren, 1989; Reiner, 2007; C. D'Antonio, personal communication, December 8, 2011; Chadden et al., 2004; F. Davis, personal communication, 2012; Marty et al., 2005.

b. Integrate riparian management into fuel management planning for upland systems.

Historically, riparian areas have not been heavily managed for fuel or fire regime goals. This is changing as it becomes clear that fire has an important role in shaping these systems (Kauffman, 2001). Because shifts in riparian fire regimes often mimic those of the surrounding upland systems, the Conservancy should consider riparian areas when planning for fire management of upland areas. For example, as fire suppression policy has increased the risk of high severity fires in conifer systems in the Western U.S., it has had similar effects on nearby riparian areas (Van de Water & North, 2011). As the science of fire management in riparian areas advances, it may be appropriate to develop detailed plans aimed specifically at riparian zones.

c. Treat high risk areas.

Under certain conditions, riparian areas can carry high intensity fires across a landscape (Van de Water & North, 2011). During droughts that follow wet years, riparian areas have high biomass and dry fuel, both of which favor extreme fire conditions. This is a particularly acute danger in areas where ignition frequencies are high, such as the portions of the Ranch adjacent to Interstate 5.

Additional areas of concern include streams overgrown by California grape (*Vitis californica*). While the literature is not clear on the danger this poses, the grape plants could potentially act as quick-burning ladder fuels (Howard, 1993).

d. Limit postfire management.

Postfire management activities such as removal of dead material and hydroseeding can be detrimental to riparian habitat (Beschta et al., 2004; Karr et al., 2004). For this reason, postfire management should be limited to approaches that allow the riparian areas to follow their natural succession. Appropriate strategies may include fencing areas off from grazing to allow for recovery, and prevention of invasive plant colonization.

e. Use best management practices for riparian systems when carrying out fire management in upland areas.

Fire management in upland areas can significantly affect riparian areas. For example, thinning can create large sediment loads and change runoff to riparian areas. Many guidelines to minimize negative effects on riparian areas have already been published (Elliot et al., 2010; Neary et al., 2005). These should be implemented on the Ranch for any fire management activity.

Riparian Systems			
Goal	Maintain mosaic of age and vegetation types; maintain natural succession of stream systems. Specific goals may vary depending on stream order.		
Monitoring Regime/Metrics	• California Stream Bioassessment • Vegetation surveys		
Mgmt Strategy	Restoration	Mechanical Thinning	Prescribed Burning
Expected Benefits	Improved vegetation; stabilization of stream slopes; reduced sedimentation; long-term benefit of increased canopy.	Reduced fuel loads; open canopy; creation of mosaic of systems.	Maintenance of nutrient cycling in the watershed; maintenance of heterogeneity of stream system and sedimentation.
Cost/acre	Varies.	$650/acre.	$45/acre.
Application of treatment	Varies depending on system.	By hand. Use existing trails before skid trails.	N/A
Where	Steep slope, altered systems, altered uplands	Steep slope, altered systems, altered uplands	Steep slope, altered systems, altered uplands
When/Season	Year round.	Spring, dependent on threatened, bird species outside wet season.	Spring, dependent on threatened, bird species
Frequency	Once + maintenance	10-15 years unless other disturbance.	10-15 years unless other disturbance.
Potential Drawbacks	Can be expensive. Without good grazing control could also be ineffective.	Increased sedimentation from removal. Increase in invasive plants from disturbance.	Severe burns could negatively impact water quality, macro invertebrate composition, and benefit invasives.
Recommendation	Recommended, in conjunction with other treatments.[18]	May be appropriate in the future.	May be appropriate in the future.

Table 4: Management strategies for riparian systems.

[18] Sources: Holl, 2007; National Park Service, 2004; Neary et al., 2005; Elliot, 2010.

3. Oak Woodlands

Key management goals for oak woodlands include preventing type conversion, promoting oak recruitment, and maintaining an overstory of mature oak trees. Careful fire management can be instrumental in achieving each of these goals, although overall ecological outcomes will be strongly influenced by other factors such as grazing and climate change. Decisions regarding fire management in oak woodlands should thus take into account the full range of stressors affecting these communities.

a. Continue fire suppression while monitoring fuel loads, stand densities, and oak recruitment.

We recommend that the Ranch continue to suppress fires in oak woodlands. Although oak woodlands are adapted to relatively frequent fires, wildland fire use entails substantial risk and unpredictability, and is therefore unlikely to be a feasible management strategy. Fire suppression should be accompanied by regular monitoring of fuel loads and stand densities. Over time, the absence of fire can lead to fuel buildup and increased stand densities (Fry, 2008; Horney et al., 2002). High fuel loads increase the risk of high-severity crown fires, which can kill mature oak trees, degrade wildlife habitat, and contribute to soil erosion (McCreary, 2004; Standiford & Adams, 1996).

Despite considerable departures from estimated pre-settlement fire return intervals in Tejon Ranch's oak woodlands, there is no evidence that fuel loads currently exceed their historical range. Livestock grazing, which is widespread in oak woodlands and is the Ranch's primary means of fuel management, can effectively control understory fuel loads (Tejon Ranch Company & Tejon Ranch Conservancy, 2009; Harrington & Kathol, 2009; Husari et al., 2006). However, future changes in the distribution or intensity of grazing, or changes in net primary production associated with climate change, could alter net rates of fuel accumulation, potentially leading to increased fire risk. It is thus important to monitor fuel loads and stand densities in Tejon's oak woodlands to ensure that undesirable and potentially hazardous fuel levels do not arise.

Lack of oak recruitment is a problem that has been observed throughout California (Tyler et al., 2008). The causes of the decline in oak recruitment are unclear, but contributing factors may include grazing, altered fire regimes, disease, acorn predation, and climate change (Tyler et al., 2006). On Tejon Ranch, recruitment rates among blue and valley oaks appear to be slightly less than rates of oak mortality on average, indicating that oak populations may be gradually declining (Appelbaum et al., 2010; Hoagland et al., 2011). There is no indication that fire suppression is responsible for this trend, but given the uncertainty around the factors controlling oak recruitment, the Conservancy should monitor recruitment rates and assess the impacts of fire management actions on oak recruitment.

b. Avoid the use of mechanical thinning or prescribed burning except where there is a compelling need.

Mechanical thinning and prescribed burning have been implemented in oak woodlands to reduce fuel loads, decrease stand densities, enhance oak regeneration, remove invasive species, improve wildlife habitat, and lessen the risk of type conversion (Fry, 2008; Fryer, 2007; Peterson & Reich, 2001; Kauffman & Martin, 1986; Holmes et al., 2011; Howard, 1992a). But mechanical thinning and prescribed burning both have significant drawbacks, and should only be applied if there is a compelling need (see Table 5). Mechanical thinning is expensive, can damage soils and vegetation, and does not fully replicate the effects of fire (Keeley et al., 2009; Husari, 2006). Prescribed burning impairs air quality and can be difficult to control; furthermore, its effects on oak savannahs and woodlands are not well-understood (Keeley et al., 2009; Fry, 2008).

There is currently no indication that active management techniques such as thinning or prescribed burning are needed in oak woodlands on Tejon. These woodlands are well-stocked with mature trees that provide valuable wildlife habitat. In valley and blue oak woodlands, grasses dominate the understory, and there is little evidence of encroachment by shrubs (Hoagland et al., 2011). Black oaks and canyon live oaks tend to be the dominant tree species where they occur on the Ranch, and thus encroachment by conifers does not appear to be a problem in these areas (M. White, personal communication, December 20, 2011).

Oak Woodlands		
Goal	Prevent type conversion, promote oak regeneration, and maintain overstory of mature oak trees.	
Monitoring Regime/Metrics	Fuel loads; tree densities; oak recruitment; tree species composition	
Mgmt Strategy	Grazing	Mechanical Thinning + Prescribed Burning
Expected Benefits	Reduction of ground fuels and mitigation of high severity wildfire risk.	Reduction of ground fuels and decreased tree density leading to mitigation of high severity wildfire risk.
Cost/acre	Maintenance of grazing infrastructure (presumably more than offset by revenues from grazing leases).	\$1700/acre. \$45/acre for prescribed burning.
Application of treatment	Allow moderate intensity grazing in oak woodlands. Use exclosures around seedlings as needed.	Mechanical thinning of younger cohorts followed by prescribed burning to decrease fuel loads.
Where	Throughout range.	Areas characterized by high fuel loads, high stand densities, encroachment by other vegetation types, or impaired oak regeneration.
When/Season	Impacts on oak seedlings may be minimized by grazing in winter.	Spring/Summer.
Frequency	Annual.	10-25 year rotation.
Potential Drawbacks	Reduced oak recruitment; soil compaction; increased spread of invasive plants.	Expense, damage to soils and vegetation, impaired air quality, risk of fire spreading out-of-control, effects poorly understood.
Recommendation	Monitor grazing impacts.	Not recommended. There is no compelling evidence that Tejon Ranch's oak woodlands are experiencing type conversion, excessive fuel loads, or other conditions that might warrant the use of mechanical thinning and prescribed burning.[19]

Table 5: Management strategies for oak woodlands.

[19] Sources: B. Kuhn, personal communication, February 7, 2012; Hoagland et al., 2011; Keeley et al., 2009; Harrington & Kathol, 2009; Fry, 2008; Husari et al., 2006; Tyler et al., 2006; Horney et al., 2002; Swiecki & Bernhardt, 1998; Holmes et al., 2011; National Park Service, 2004.

4. Montane Conifer Forests

a. Survey conifer forest structure and composition.

Before any active management treatments are considered, the current forest structure and fuel loads of Tejon's conifer stands should be carefully surveyed. Carrying out this on-site 'needs assessment' is critical for fully understanding how the forest structure of stands on the Ranch may or may not be creating dangerous fuel dynamics. Overly dense conditions can exacerbate drought, disease, and insect outbreaks, and heighten the risk of stand-replacing crown fires. If surveys suggest that the forest structure and fuel loads in Tejon's conifers are in fact at unhealthy levels, then active thinning treatments may be considered (see Table 6). Such fuel management actions can promote the vitality of the system by mitigating fire risk, and encouraging the general health and resilience of conifer forests (Ritchie et al., 2007).

b. Thinning can reduce dangerous fuel dynamics.

If initial surveys indicate the need for management intervention, then mechanical and hand thinning can be used to target trees of specific size cohorts. Thinning treatments can reduce dangerous fuels and relieve forest stress due to crowded conditions, thereby promoting an open forest composed of mature trees (Ritchie et al., 2007). Thinning can also allow more light to reach the forest floor and encourage recruitment of shade intolerant yellow pines (Habeck, 1992). Hand thinning with chainsaws can remove smaller trees up to about 10-14" dbh while mechanical thinning with heavy equipment is needed for removal of larger trees (Holl, 2007). Focusing operations on fire-prone areas, such as dry south-facing slopes and ridgetops, can minimize costs by reducing the size of the treatment area. This strategy can also result in varied age classes, and create low-fuel tracts that are less likely to carry a fire (Schmidt et al., 2008). An approach the Conservancy may find beneficial is to initially employ hand-thinning treatments that are less intrusive and less expensive. If monitoring suggests that desired density, fuel reduction, and forest health benefits are not being achieved, then mechanized thinning may be needed.

c. Thinning can be combined with prescribed burning.

A combined treatment approach where thinning is followed by a prescribed fire offers the most significant fuel and fire risk reductions, and may cause forests to more closely resemble presettlement conditions (Agee & Skinner, 2005; Stephens, 2005). Fire is not necessary for the propagation of Tejon's conifer species, but it can aid early growth (Habeck, 1992). Conducting prescribed burns in the spring (when fuels are still moist) and/or shortly after the thinning treatment can make fires less intense and easier to control. This will also facilitate retention of duff and snags for improved wildlife habitat conditions (Weatherspoon et al., 1992). In addition, prescribed burns can "sanitize" the forest against unwanted insects and diseases (Zouhar, 2001). For these reasons, prescribed burning used in conjunction with mechanical thinning may be appropriate, particularly if monitoring indicates that hand thinning is not achieving desired goals for density and fuel reduction, or if drought or disease outbreaks make additional intervention necessary.

It is important to note, however, that we do *not* recommend prescribed burning without mechanical pretreatment. In a worst-case scenario, the fire could spread out of control and become a large-scale crown blaze capable of killing extensive tracts of mature trees (Verner et al., 1992). This could open the forest to invasion by herbaceous species and potentially cause a phase shift in the ecosystem (Goforth & Minnich, 2008). Moreover, intense, smoldering ground fires could kill large trees by damaging roots, and fire wounds could make the forest more susceptible to insect and disease infestations (Taylor, 2000). White fir in particular tends to be less fire resistant than shade intolerant yellow pines, even when mature[20] (Zouhar, 2001).

d. Exclude Grazing Activity

Livestock grazing in conifer systems has been shown to greatly increase the density of conifer seedlings. This harms forests by increasing competition and promoting fuel accumulation (Belsky & Blumenthal, 1997). Excluding livestock from conifer systems on Tejon with barbed wire fencing will likely reduce conifer density, improve forest health, and minimize fire risk. To test the efficacy of this strategy on the Ranch, the Conservancy may wish to fence off particular conifer stands or experiment with fenced and un-fenced plots in conifer tracts (See Figure 17 for associated installation costs).

[20] Studies have found mixed effects on white fir stands, with some research showing declines of white fir relative to other species, and others finding that white fir retained its dominance (Van Mantgem et al., 2011; North et al., 2007; Schmidt et al., 2006; Fule et al., 2004; Mutch & Parsons, 1998).

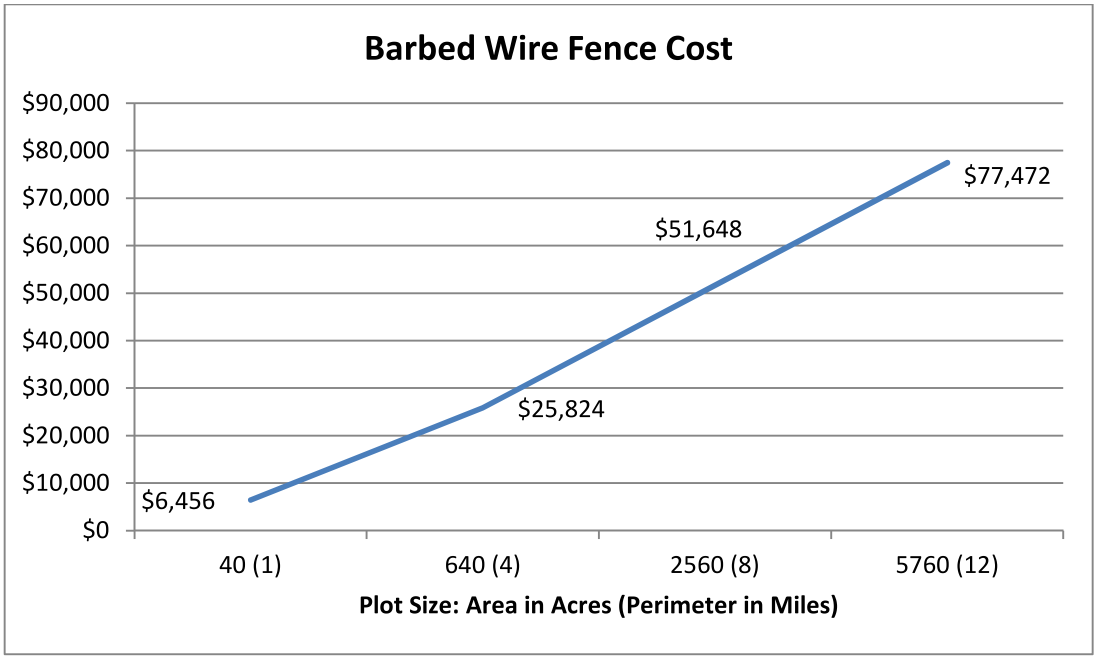

Figure 17: Cost of various sizes of barbed wire fence plots.

Mixed Conifer Forest			
Goals	• Prevent type conversion; minimize risk of crown fire • Survey current forest structure and composition		
Monitoring Regime / Metrics (Criteria)	• Forest density basal area maintained between 28-34 square meters/ha or ~12–70 mature trees/ha • Average fuel loads maintained between 25-100 metric tons per hectare		
Mgmt Strategy	Exclusion of grazing	Hand thinning	Hand thinning + prescribed burning
Expected Benefits	Reduced seedling density and ladder fuels; improved forest health.	Reduced density, ladder fuels and fire risk; improved forest health.	Reduced density, ladder fuels and ground fuels; improved forest health; greatest reduction to fire risk.
Treatment Cost	$1,614/quarter mile barbed wire fence.	$650/acre.	$950/acre.
Application of treatment	Installation of barbed wire fencing.	Hand thinning of trees less than 10" dbh and less than 11 feet tall.	Hand removal of trees less than 10" dbh and 11 feet tall, followed by prescribed burn.
Where	Particularly dense stands subject to heavy grazing.	High risk areas: ridgetops and southern slopes.	High risk areas: ridgetops and southern slopes.
When/Season	Any.	Any.	Spring.
Frequency	Single installation.	Dependent on monitoring.	Dependent on monitoring.
Potential Drawbacks	Slight loss of grazing area for cattle.	Accumulation of cut fuel on forest floor; wounding of residual trees; insect and disease invasion.	Fire damage to non-target trees, tree wounding and consequent disease and insect susceptibility.
Recommendation	Recommended.	Recommended.	Recommended, provided that stands are pre-thinned.[21]

Table 6: Management strategies for montane conifer systems.

[21] Sources: Schmidt et al., 2008; Holl, 2007; North et al., 2007; Ritchie et al., 2007; Mayer, 2005; Stephens, 2005; Zouhar, 2001; Belsky & Blumenthal, 1997.

5. Chaparral

a. Fire suppression should continue.

We recommend continued fire suppression in the Ranch's chaparral areas (see Table 7). If fires were allowed to burn through chaparral vegetation, they could easily escape containment and pass into neighboring ecosystems where fire may have more severe impacts. Given the patchy nature of chaparral stands along the Tehachapi Mountain Range and in the southern portion of the Ranch, fires may quickly spread from chaparral into neighboring vegetation. Unpredictable wind patterns, the occurrence of chaparral vegetation along steep slopes, and the flammable nature of chaparral fuels may combine to produce erratic fire behavior and rapid rates of spread, making containment difficult.

b. Fuel treatments are not needed in wildland areas.

While fires in chaparral should be suppressed to prevent uncontrollable fire behavior, some stands may benefit from the occurrence of fire. As long as stands reach maturity before fire occurs (a period of two decades for most species), fire is generally beneficial. An estimated 12,000 acres, or 75%, of chaparral on the Ranch is over 20 years in age. Of these mature stands, over 25% are likely over 100 years in age. Because these mature stands would not be adversely impacted by fire, fuel treatments intended to inhibit fire spread through chaparral vegetation have little value in wildland areas on the Ranch.

Prescribed burns have been justified in chaparral for enhancing natural resource values (Keeley, 2002). Yet there is little evidence that aging stands will significantly degrade in the long term absence of fire (Keeley, 1992a; Keeley, 1992b). Furthermore, total exclusion of fire in chaparral is unlikely to be possible despite best efforts to control fire spread. Within the past twenty years, for example, approximately 4,200 acres of chaparral have burned on the Ranch in spite of suppression efforts. This is consistent with regional trends. Studies have shown that fire suppression has failed to exclude fire from southern California shrublands and that more area has burned since the advent of active fire suppression (Keeley, 2002). The combination of older stands exhibiting little indication of decline and the likely occurrence of some fire in spite of active suppression suggests that prescribed fire is not needed to enhance stand health on the Ranch.

Prescribed burns could in theory be justified to enhance natural resource values if the majority of the Ranch's chaparral was fire-free for so long that stands became uniformly mature and exceeded the upper range of their natural fire interval. If all the stands of chaparral on the Ranch were to escape fire for a century or more, then those species which remain competitive in the long-term absence of fire would likely out-compete species reliant on fire for germination (Keeley & Zedler, 1978). But this is neither the current condition, nor a likely future. As long as the majority of stands do not exceed the upper range of their natural fire interval (80-100 years), chaparral is best left untreated.

Another common motivation for using prescribed fire in chaparral is wildfire mitigation (Keeley, 2002). Landscape-scale rotational burning has been justified in chaparral as a method for creating mosaics of differentially aged stands which may naturally inhibit fire spread. However, fire has been shown to burn through younger age classes in extreme fire weather (Conard & Weise, 1998). Thus, prescribed fire as a means of achieving reduced fire hazard is limited to moderate weather conditions. Keeley (2002) proposes that fuel treatments are most advantageous in select locations which are susceptible to high frequency ignitions, such as near the urban wildland interface. On the Ranch, fires are most frequent alongside major transportation corridors. County and state fire agencies take the lead on fuel treatments in these areas.

Chaparral		
Goals	Reduce risk of uncontainable wildfires. Prevent loss of shrub species diversity.	
Monitoring Regime / Metrics	• A diversity of differently-aged stands. • Occurrence of both seeding and sprouting shrub species.	
Mgmt Strategy	Active fire suppression	Prescribed fire
Expected Benefits	Fire regimes closer to natural frequencies than would occur without suppression; chaparral fuels that do not contribute to catastrophic fires; protection of obligate seeding species; prevention of type conversion when stands are less than 10-20 years in age.	Increased germination of dormant seedbanks and protection of obligate seeding species; may limit fire spread under moderate weather conditions, though is not expected to inhibit fire spread under severe weather.
Cost/acre	No additional cost to Conservancy.	~$31/acre (2004 estimate for a 500 acre burn on a 5% slope).
Application of treatment	Suppress any fire that starts in, or spreads to, chaparral vegetation.	Only applicable if the majority of stands escape fire for a prolonged period of time, producing stands that are homogenous age and greater than 80 to 100 old.
Where	Where chaparral vegetation occurs.	Stands that contain obligate seeding species and are greater than 80 to 100 years old.
When/Season	Year-round.	Winter or spring, when fire weather is less severe. Winter and spring burns may also be less beneficial to invasives than fall burns.
Frequency	As needed.	Infrequently, if at all.
Potential Drawbacks	Active fire suppression may delay fire-stimulated seed germination in older stands.	Spring prescribed fires may adversely impact nesting wildlife; short-term reduction in air quality; risk of fires escaping control.
Recommendation	Recommended.	Not recommended.[22]

Table 7: Management strategies for chaparral communities.

[22] Sources: Potts & Stephens, 2009; Erickson & White, 2007; Keeley, 2006b; USDA, 2004; Keeley & Fotheringham, 2001; Conard & Weise, 1998; Zedler, 1995; Keeley, 1992a; Keeley, 1992b.

6. Joshua Tree Woodlands and Desert Scrub

a. Fire suppression should continue.

The greatest threat to the Ranch's Joshua tree woodlands and desert scrub communities is type conversion due to the invasive/wildfire cycle (Brooks et al., 2011; Brooks & Matchett, 2006; DeFalco et al., 2009). Because fire aids invasive grasses at the expense of native vegetation, has been shown to harm wildlife such as the desert tortoise, and is not needed by any native species (including *Yucca brevifolia herbertii*), we recommend continuing fire suppression in the Ranch's deserts (see Table 8).[23]

b. The Conservancy should monitor ground cover of invasive annual grasses.

While there is little quantitative data on the relationship between invasive annual grass cover and fire risk in desert scrub, a correlation between *Bromus tectorum* ground cover and fire risk has been found elsewhere.[24] Extrapolation from a 2006 study of *Bromus tectorum* cover and ignition probability[25] in sagebrush steppe predicted that fire risk would be less than 30% if *Bromus* cover were limited to 5%

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[23] Due to the threat of soil disturbance—compacted desert soil can take over a century to return to its previous condition, and erosion of even a few centimeters can lead to "[g]ross disorganization of community structure"—we do not recommend mechanical thinning (Webb, 2002; Lovich & Bainbridge, 1999; Webb & Stielstra, 1979).

[24] Moreover, invasive grasses thrive on disturbances such as grazing, which is present on the Ranch (U.S. Fish and Wildlife Service, 2011b; Brooks et al., 2011; Brooks & Matchett, 2006; Webb, 2002; Brooks, 2000; Lovich & Bainbridge, 1999; Webb & Stielstra, 1979).

[25] The study mapped ground cover of *Bromus tectorum* using aerial photography at a time of year when dead *Bromus* could be distinguished from other vegetation by color, and defined fire risk as the chance that a single ignition attempt at a random point would lead to a fire of at least 100 m^2 (Link et al. 2006).

(Link et al., 2006). While cover percentages observed in sagebrush steppe are unlikely to correlate precisely with fire risk in sparsely-vegetated desert scrub, qualitative observations of the invasive/wildfire cycle in deserts suggest that grass cover and fire risk are related (DeFalco et al., 2009; Brooks & Matchett, 2006).

The Conservancy could develop a quantitative understanding of this relationship by replicating the Link et al. (2006) study in the Ranch's deserts. This would entail significant risks, however, including the prospect of a fire burning out of control. A safer but less informative course of action would be to conduct annual surveys of invasive grass cover, and observe trends over time. If future aerial surveys of Joshua tree distribution[26] are conducted at the beginning of the dry season (when invasive annual grasses are visible by color), flights could be extended to Mojavean scrub areas, and the resulting photographs could provide a rough measure of invasive grass cover (Appelbaum et al., 2010; Zouhar, 2003). The Conservancy could also use ground-level point quadrats, although extensive sampling may contribute to soil compaction. Whether obtained by aerial survey or point quadrat, however, periodic measurements of invasive grass cover could indicate the extent to which the invasive/wildfire cycle is advancing.

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[26] The Conservancy should continue to monitor Joshua tree distribution, health and height, as well as desert night lizard (*Xantusia vigilis*) presence, in accordance with the recommendations of Appelbaum et al. (2010).

Joshua Tree Woodlands and Mojavean Scrub	
Goal	Avoid type conversion by slowing or stopping the invasive/wildfire cycle.
Monitoring Regime/Metrics	• Acres of Joshua tree woodland or Mojavean scrub burned
Mgmt Strategy	Active fire suppression.
Expected Benefits	Reduced disturbance; dampening of invasive/wildfire cycle.
Cost/acre	No additional cost to Conservancy.
Application of treatment	Suppress any fire that starts in, or spreads to, deserts.
Where	Throughout Ranch's deserts.
When/Season	Year-round.
Frequency	Only needed during fire event.
Potential Drawbacks	N/A.
Recommendation	Recommended.[27]

Table 8: Management strategies for Joshua tree woodlands and Mojavean scrub.

VI. RECOMMENDATIONS FOR FUTURE RESEARCH

The recommendations made in this Report are based on conclusions drawn from available data, such as the Ranchwide vegetation survey, aerial imagery, and fire perimeter records. These sources provide only coarse approximations of vegetation structure, composition, and age, among other factors related to fire management on the Ranch. Additional research and field surveys are therefore necessary to validate certain assumptions made in this Report. For example, the extent and continuity of invasions by fire regime-altering species, such as cheatgrass or tamarisk, should be surveyed and monitored along with desired species, such as

[27] Sources: U.S. Fish and Wildlife Service, 2011b; Appelbaum et al., 2010; DeFalco et al., 2009; Brooks & Matchett 2006; Webb, 2002; Brittingham & Walker, 2000; Lovich & Bainbridge, 1999; Webb & Stielstra, 1979.

native forbs and grasses, to better understand how current management regimes are affecting establishment patterns and species viability.

Livestock grazing is widespread on the Ranch, and is likely to be one of the primary drivers of fire regimes. It affects them by reducing the abundance and continuity of fine fuels and by changing the structure and composition of vegetation. Although grazing serves as a surrogate for fire in terms of reducing fuel loads, grazing and fire have different ecological effects and influence fire regimes in different ways. Key uncertainties include the impacts of grazing on native vs. nonnative grasses, and the impacts of grazing on hardwood recruitment. Each of these effects will likely depend on the intensity, duration, and timing of grazing. We recommend additional research to address these uncertainties and improve understanding of the interactions between grazing and fire regimes.

Computer modeling is a valuable tool for understanding the potential impacts of climate change, development, and alternative management strategies on the Ranch's fire regimes. Our LANDIS-II analysis could be expanded to further explore the three questions we examined or to explore additional questions relevant to fire management. Some steps that could be taken include 1) running additional replicates of each scenario to increase the statistical power of the results, 2) refining species input parameters (such as age cohorts, establishment probabilities, or maximum net primary production) based on future research and surveys, 3) using newer GCM predictions to simulate the effects of climate change, 4) modeling a larger area of the

Ranch, 5) conducting a thorough sensitivity analysis, and 6) testing alternative management scenarios (varying management areas and/or management treatments).

VII. CONCLUSION

Our analysis suggests that fire regimes on the Ranch may be changing. Fires on the Ranch have become more numerous in recent years and, unlike in the region as a whole, they may be increasing in size. The FRID maps reveal that this change is unevenly distributed: even with greater numbers of fires in recent decades, parts of the Ranch are burning less frequently than historical norms. As climate change intensifies, and continued development leads to more anthropogenic ignitions, the Ranch's fire regimes may continue to shift.

Understanding these changes is likely to require improved monitoring of fuel conditions in the Ranch's major ecological communities. With the exception of fire suppression, which appears to be appropriate across the Ranch, adaptation measures will vary by community. In certain communities, such as montane conifer systems, active management may be necessary to prevent type conversion.

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APPENDIX A: BACKGROUND ON THE RANCH'S ECOLOGICAL COMMUNITIES

1. Grasslands

a. Drivers and Background Fire Regimes

Fire behavior in grasslands is driven by fuel structure, moisture, vegetation composition, topography, and weather (Reiner, 2007). The type, structure, and continuity of fuel, along with the relatively flat topography of grasslands, provide few barriers to air movement and create a fire-prone environment. Fuel moisture tends to decline quickly in grasslands, also favoring frequent fires (Vogl, 1979). Knapp (1998) identified a link between large fires in intermountain rangelands and large amounts of nonnative grass fuel. However, Keeley et al. (2011) note that fires have been observed to spread even in grazed areas, suggesting that the influence of wind could override fuel structure in some cases. The mountain ranges bordering the Central Valley block the influx of foehn winds coming from the west, but powerful winds sometimes originate from north of the Valley. In the Central Valley, fire size may be augmented by temperature inversions that expose fires to warmer air (Wills, 2006). Topography also influences fire behavior in grasslands, and increases in slope will translate to faster fire movement (Biswell, 1989). Moreover, vegetation type and continuity may interact with topography to direct fire outcomes, as Knapp (1998) found that heavily-invaded areas with level topography tended to support large fires in the Intermountain West.

Prior to European settlement, fires likely occurred at short intervals as a result of the combustible nature of grass fuels and anthropogenic ignitions[28] (Keeley et al., 2011). An estimate based on accounts of Native American burning by Stephens et al. (2007) places the pre-European median FRI at 3 years and the high FRI at 8 years for California steppe vegetation communities. In other desert grasslands in the southwest outside of California, fire is thought to have occurred regularly during pre-European times, with a frequency as brief as every ten years (McPherson, 1995). Lightning-caused ignitions within the Central Valley were probably uncommon, as only a few lightening strikes per year tend to occur per 100 km^2, but the adjacent higher elevation regions could have provided a source of fire (Wills, 2006).

It has been argued that, within the Central Valley, invasives have extended the fire season and made grasslands more fire-prone (Wills, 2006), potentially to the detriment of native perennials. Fires move more slowly and at lower temperatures in

[28] Native Americans used fire to preserve grasslands and open forests, which provided them with plant food, building materials, and small game. Fire was also used to convert other communities to grasslands (Stromberg et al., 2007).

grasslands not overrun with nonnative annual grasses (Reiner, 2007), which are believed to have altered grassland fuel structure by decreasing fuel fragmentation (Wills, 2006). However, a lack of historical data makes it impossible to definitively describe the impact of invasive grasses on fire regimes in these ecosystems (E. Allen, as cited in Chadden et al., 2004).

b. Effects of Fire

Fire induces nutrient cycling and increases nitrification in grasslands, but it has been argued that plants are more impacted by changes in light and temperature associated with fire (Boerner, 1982). Ash and residual organic substances contribute to the soil's ability to retain water and provide habitat for microorganisms (Vogl, 1979). However, volatilization of nutrients also occurs (Boerner, 1982), and burning on short intervals is contraindicated by studies showing that large quantities of sulfur and nitrogen can be lost as a result (D'Antonio et al., 2002; Menke, 1992).

Vegetation impacts from fire include grass and forb mortality, bunchgrass top-kill and fragmentation, loss of aboveground seeds, and germination of some invasive and native forb species' seeds (Reiner, 2007). Fires are more damaging to woody species, and thus help to prevent the conversion of grasslands to other vegetation communities. Fire can benefit bunchgrasses and other species by removing growth-inhibiting litter (Vogl, 1979), and fully-grown bunchgrasses can also be subject to "self-shading" if some plant material is not removed (Dyer, 2003). For example, a study in Northern California found that legumes and broadleaf species, many of which were native, responded positively to mulch removal (Heady, 1956).

Significant postfire growth has been observed in some native perennial species, including *N. pulchra* (Wills, 2006). Dyer (2003) reported the increased growth of mature *N. pulchra* as a result of 7 years of prescribed fire and grazing, though average basal area initially dropped in the first year. The impact of multiple burns may not be represented by the bunchgrass damage often reported after a single burn, which can result from hot fires generated by the built-up of dead plant material (Dyer, 2003). In contrast, higher *N. pulchra* mortality subsequent to prescribed burning has also been reported, theorized in one case to be the result of high fuel levels (Marty et al., 2005). It has been argued that fires of higher intensity in particular may be damaging to bunchgrasses (Reiner, 2007). However, a higher number of seedlings have been observed to follow this response (Gillespie & Allen, 2004; Marty et al., 2005). Perennial seed generation also initially declines as a result of prescribed burning, but this effect is limited to the first year (Menke, 1992). In addition, Dyer (2002) observed an increase in the size of *N. pulchra* seeds and a 72% increase in germination probability 10 years after prescribed fire. A lack of response to fire has also been reported for perennial bunchgrasses. In a meta-analysis, D'Antonio et al. (2002) determined that fire may not result in *N. pulchra* increases. In California coastal grassland, Hatch et al. (1999) found that *N. lepida and N.*

pulchra did not exhibit statistically significant differences in cover or frequency in the year after fall prescribed burning. Similarly, two years of fall prescribed burning in coastal California had no impact on perennial grasses including *N. lepida and N. pulchra* (Bartolome et al., 2004).

The native annual *Vulpia microstachys* is one of the primary annual grass species in the Antelope Valley grasslands on the Ranch (Bartolome et al., 2010-2011). After fall burning in mixed woodland and chaparral in Oregon, *V. microstachys* exhibited either a positive or neutral response (Coulter et al., 2010). Prescribed fire in shrub-steppe in Washington during the fall also failed to have any impact on *V. microstachys* (Link & Hill, n.d.). One study in California found that the type of soil present may change the way native species such as *V. microstachys* respond to fire (Harrison et al., 2003).

Grassland fire is associated with increased forb populations (Harrison et al., 2003; Keeley et al., 2011; D'Antonio et al., 2002; Pollak & Kan, 1998; Hastings & DiTomaso, 1996; Meyer & Schiffman, 1999; Parsons & Stohlgren, 1989; Dickens et al., 2008; Wills, 2006; Reiner, 2007; Gillespie & Allen, 2004; Hervey, 1949). Forb seeds respond to the removal of thatch (D'Antonio et al., 2002) and competitive nonnative grasses resulting from fire (Dickens et al., 2008). For instance, Hervey (1949) found that the proportion of forbs in annual-dominated grassland in coastal California rose in the year after prescribed fire, such that 45% more of the vegetation density on burned sites was comprised of forbs. In another example, fire diminished nonnative grasses that were observed to compete with native *E. macrophyllum* more than native grasses, likely supporting enhanced *E. macrophyllum* reproduction, though burning may also lead to fewer *E. macrophyllum* seedlings (Gillespie & Allen, 2004). Among the forbs that benefit from fire are nonnative forb species (Harrison et al., 2003; Gillespie & Allen, 2004; Keeley et al., 2011; D'Antonio et al., 2002; Pollak & Kan, 1998; Parsons & Stohlgren, 1989; Dickens et al., 2008; Reiner, 2007). In one study, fire in the Santa Rosa Plateau increased nonnative forb cover (primarily *Erodium botrys*) while leaving native forbs in general unchanged, possibly as a result of low water availability or few native forb seeds (Gillespie & Allen, 2004).

Harrison et al. (2003) observed a higher number of native and nonnative grassland plant species after a fall wildfire, with richness of native species growing most in serpentine soils within a year of burning. Nonnative species experienced the largest overall increase in richness, with the greatest increases in more heavily-invaded nonserpentine soils. The authors surmise that soil characteristics may mediate the impact of fire and grazing, and that fire may augment nonnative plant diversity in alien-dominated ecosystems.

Fire has been employed to reduce nonnative species in California grasslands. Fires may help to eliminate the seeds of nonnative annual grasses if they are subjected to burning before seeds are incorporated into the soil (Pollak & Kan, 1998). In late spring, perennial species can rebound even if seeds are burned, because of their tendency to respond to fire with sprouting (Keeley, 2001). For example, Pollak and Kan (1998) observed a reduction in nonnative grass cover and an increase in native grass and forb cover one year after a June prescribed burn at Jepson Prairie in California. This result may be due to a release from competition for resources, including space and water, as well as a reduction in thatch. Similarly, invasive grass cover was reduced by 35.4% after experimental fire during the spring in the Santa Rosa Plateau (Gillespie & Allen, 2004). However, frequent fires tend to select for annuals that generate large numbers of seeds (Reiner, 2007).

Research in blackbrush (*Coleogyne ramosissima*) communities suggests that, under Mojave Desert climatic conditions, grass and herbaceous species may persist and even expand with fire, but nonnative species may be favored (Brooks & Matchett, 2003). There is also some indication that fire could offer some control of *Bromus rubens* and encourage native perennial forbs within desert systems (Abella et al., 2009), but significant reductions of *B. rubens* may not always be observed (Brooks & Matchett, 2003), and this result may not be generalizable to grasslands.

c. Changing Fire Regimes

Modeling studies suggest that climate change will affect grassland fire regimes. Lenihan et al. (2008) found that grasses became more prevalent with changes in moisture under multiple projections of climate change (GFDL-A2, GFDL-B1, and PCM-A2 scenarios), leading to larger fires and further contributing to the recession of other vegetation communities. The only exception was found in the deserts under the GFDL scenarios, in which grasslands near desert systems were overtaken. Fried et al. (2004) predicted more grassland area to be impacted by fires in northern California under a GISS GCM double carbon dioxide scenario when results from a model incorporating fire suppression were generalized from Humboldt,[29] Amador, and Santa Clara CDF ranger units. This considered only the size of fires successfully controlled. Fires in grasslands tended to be more frequent and to move across the landscape more rapidly. It was also determined that more grass fires would become uncontrollable, especially in areas of low population density, because of impacts on fire suppression efforts.

d. Management Approaches

1) Grazing

Grazing occurs throughout the Ranch's grasslands, and has complex, differential effects on community composition (D'Antonio et al., 2002; Huntsinger et

[29] In this and similar CDF units, however, climate change actually *reduced* the amount of grassland impacted by fire.

al., 2007; Seabloom et al., 2003; Stahlheber & D'Antonio, 2011). According to a meta-analysis of grazing studies in California grasslands, grazing increases species cover and richness among native forbs, but also increases species richness among invasive grasses[30] (see Table 9) (Stahlheber & D'Antonio, 2011).

	Native	**Invasive**
Grasses	Species cover: Increased in response to wet-season grazing; slightly increased in response to continuous grazing; unaffected by dry season grazing. Species richness: Unaffected, but many sites had only *N. pulchra*.	Species cover: Decreased in response to wet-season grazing; unaffected by continuous or dry-season grazing. Species richness: Increased.
Forbs	Species cover: Inconsistent effects statewide, but consistently increased in interior (i.e., non-coastal) grasslands. Species richness: Increased.	Species cover: Increased cover at wetter sites; no effect or declining cover at arid sites. Species richness: Unaffected.

Table 9: Effects of grazing on species cover and species richness of native grasses, native forbs, invasive grasses and invasive forbs (Stahlheber & D'Antonio 2011).

In some cases, grazing has been reported to harm native species. One study determined that grazed grassland on the Carrizo Plain was associated with fewer native species and the complete absence of bunchgrasses, unlike a site that had not been grazed (Kimball & Schiffman, 2003). Forb diversity has also been observed to be greater in an ungrazed portion of the Carrizo Plain. The Carrizo Plain is a fairly dry landscape, which may explain why, unlike elsewhere in California, grazing there has been found to be severely detrimental to native species (P. Schiffman, personal communication, January 30, 2012). Fewer native species and more nonnatives have been observed with grazing on nonserpentine soils in northern California, though not on serpentine soils. Like fire, grazing may enhance nonnative species where they already dominate (Harrison et al., 2003).

[30] At least one of the studies on which this conclusion was based, however, was conducted in coastal prairie, not the drier, interior grasslands of Tejon Ranch (Hayes & Holl, 2003). Moreover, neither grazing nor the cessation of grazing can wholly reshape communities that are already dominated by invasives. As Seabloom et al. (2003) point out, certain native annual forbs may be seed-limited, and therefore unable to take full advantage of gaps in cover created by grazing.

Research suggests that the impacts of fire and grazing on California grasslands may not be interchangeable. Specifically, Marty et al. (2005) found that *N. pulchra* culm formation was inhibited by grazing, though the number of new plants in the population did not seem to drop. Unlike fire, grazing was not associated with the death of plants, but also did not increase the number of seedlings observed two years after fire.

In some cases, natives benefit from or are unharmed by grazing. For example, *N. pulchra* died off more frequently in areas not subject to sheep grazing at Jepson Prairie, though this relationship was not always statistically significant (Dyer, 2003). Menke (1992) points to research suggesting that the removal of nonnative grass seeds and an increase in light to bunchgrass tillers and seedlings can be achieved by early spring high intensity grazing, while high intensity grazing in the summer can also eliminate dead plant material that blocks light.

Native perennial grasses have been observed to increase in cover after wet season grazing, but not after dry season grazing (Stahlheber & D'Antonio, 2011). This may be a result of reduced competition from invasive annuals, whose phenology renders them more vulnerable to early spring grazing[31] (Huntsinger et al., 2007). In some cases, however, the results of grazing may be difficult to separate from the effects of abiotic factors such as soil and climate (Stahlheber & D'Antonio, 2011; Osem et al., 2002).

Some studies indicate that *N. pulchra* may be promoted by a combination of grazing and fire. These findings may be relevant to the bunchgrasses found on the Ranch. Dyer (2002) reported that in clearing away dead vegetation, grazing likely resulted in cooler fires during the study, thereby permitting bunchgrasses to respond positively to the burning. Enhanced growth of *N. pulchra* as a consequence of livestock herbivory and burning has been detected (Dyer, 2003). Probability of seed germination and seed size have been shown to be greater for grazed and burned *N. pulchra* plants when compared to plants that have been subject to grazing or burning alone (Dyer, 2002). In contrast to the positive results described in other studies,

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[31] By manipulating timing and animal type, managers can also use grazing to target other invasives. For example, yellow star thistle (*Centaurea solstitialis*) is highly palatable to livestock during the bolting stage, and intensive grazing can significantly affect its growth and seed production. As it reaches maturity, it becomes spiny and unpalatable to cattle, but remains palatable to goats (Huntsinger et al., 2007). Animal type can also be manipulated to control the evenness of fuel reduction. Brahma and zebu cattle use less water, and are better adapted to hot weather and hilly terrain, than the more common Hereford and Angus cattle. For this reason, instead of staying near water (and therefore having a disproportionate impact on riparian areas), Brahma and zebu cattle may reduce fuel loads more evenly across the Ranch's grasslands and savannahs (Huntsinger et al., 2007).

however, Marty et al. (2005) reported that varying grazing intensity and fire yielded the same results in all but one case and that no significant differences related to reproduction or mortality of *N. pulchra* were found. Grazing may also mitigate the proliferation of nonnative forbs after fire (D'Antonio et al., 2002), and may have a stronger adverse impact on invasive species after burning than it does at other times (C. D'Antonio, personal communication 2011).

In addition to its effects on community composition, grazing may impact wildlife. According to Germano et al. (2001), southern San Joaquin Valley vertebrates such as the giant kangaroo rat (*Dipodomys ingens*), the San Joaquin antelope squirrel (*Ammosperophilus nelsoni*), and the blunt-nosed leopard lizard (*Gambelia sila*)—all of which are threatened or endangered species found on the Ranch—are adapted not for the dense grasslands that characterize this part of the Ranch now, but for desert saltbush scrub. Pointing to several studies questioning the previous classification of the southern San Joaquin Valley as a bunchgrass prairie, Germano et al. argue that the animals of this region evolved in conditions of sparse ground cover (Germano et al., 2001). A key reason for the decline of species such as the kangaroo rat and antelope squirrel, they posit, is dense vegetation cover that interferes with foraging and seed caching, and may lead to predation by better-adapted species. Grazing may therefore be necessary to reduce cover levels during the growing season (Germano et al., 2001). The Conservancy could investigate this hypothesis by measuring kangaroo rat, antelope squirrel and blunt-nosed leopard lizard abundance in otherwise-similar grazed and ungrazed plots.

Grazing may also affect the nitrogen cycle. If ungrazed grasslands burn more frequently, they can suffer pyrodenitrification; if they do not burn, the accumulated dead matter may reduce nitrogen fixation by blocking new primary productivity (Jackson et al., 2006). Moderate grazing may therefore promote nitrogen retention, though intense grazing can lead to rapid cycling and losses through leaching (Holdo et al., 2007).

2) Mechanical Thinning

Hand thinning may be used to remove fire-regime-changing invasives on a small scale, although this approach is very labor intensive (Hoshovsk & Randall, n.d.). The success of hand thinning is dependent upon the removal of a plant's growing points. For this reason, hand removal of annual and biennial species (where a cut a few inches below the soil surface is sufficient) is typically more effective than hand removal of creeping perennials, which send out both vertical and lateral roots and rhizomes. Perennials typically regenerate from adventitious buds on deep lateral and vertical roots (DiTomaso et al., 2007).

Mechanical control by cutting or mowing is of limited use in grasslands because it causes significant soil disturbance, potentially favoring invasive species

(Erickson & White, 2007). Mowing does not control cheatgrass, because cut stems regenerate new culms and produce seeds at the cut height. Stems that are cut after seed ripening will die, but by this point the seeds are already viable in the soil. Thus, regardless of when mowing occurs, cheatgrass is likely to reestablish. Repeated mowing (every three weeks) may eliminate cheatgrass seed production but is of little relevance for wildland management (Carpenter & Murray, 1999).

3) Prescribed Fire

Prescribed fire is employed in grasslands to suppress nonnative species and promote native species. For example, prescribed burning is applied within the Santa Rosa Plateau Ecological Reserve's grassland and oak habitats every 5 to 7 years (California Department of Forestry and Fire Protection Riverside Unit, 2009). Declines in nonnative grasses have been documented, though repeat burns may be necessary to sustain reductions (Dickens et al., 2008). Recovery of invasive grasses has been observed in as short as two years (Meyer & Schiffman, 1999). Prescribed burns on a 3 to 4 year cycle have been suggested in general for grasslands (Menke, 1992).

Seasonality of fire is thought to play a role in determining the impact of prescribed burning on native and nonnative species. Late spring has been suggested as a good time for burning, as it eliminates undispersed seeds, negatively impacting invasive grass populations (Pollak and Kan, 1998). Further support for late spring fire is provided by Meyer and Schiffman (1999), who found that winter burns in the Carrizo Plain targeting invasive grass seedlings produced the lowest intensity fire and had no effect on native species cover and diversity, while burning in the late spring and fall had a positive effect on natives, especially annual forbs. These results may be particularly relevant to Tejon because the Carrizo Plain represents a relatively arid California grassland. Meyer and Schiffman (1999) also emphasize the importance of the timing of invasive seed development in scheduling burns.

Though fire may benefit some species, prescribed burning in grasslands has several drawbacks. Multiple burns may be necessary to achieve objectives, and this may severely reduce grassland nutrient stores (C. D'Antonio, personal communication, December 8, 2011), while enhancing the growth of invasive species that respond to more frequent fire (DiTomaso et al., 2006). Additional drawbacks include unwanted fire spread and air quality concerns (Menke, 1992). Many managers cite air quality impacts as one of the major barriers to burning (Chadden et al., 2004). Moreover, it is often difficult to achieve the right burn conditions, including adequate biomass over multiple burns (DiTomaso et al., 2007) to fuel a fire hot enough to kill the seedbanks of nonnative species and skew composition toward native species (C. D'Antonio, personal communication, December 8, 2011).

4) Herbicides

Herbicides can be a relatively cost-effective way to control species that adversely impact grassland fire regimes. Most herbicides should be applied in the fall or winter to prevent germination of cheatgrass and other nonnative annuals in the spring. If herbicides are applied after a prescribed burn, they should be applied to soils before seeded perennial grasses emerge in order to remove any invasive species that remain in the seedbank (Carpenter & Murray, 1999).

Despite some successes, lasting control of undesired grassland species with herbicides is an area of continuing research. Control of invasive annual grasses does not ensure that desired native species will be able to reestablish in treatment areas. In some cases, invasive species may be better able to take advantage of openings created by herbicide application (Allen et al., 2005). In such instances, herbicides may be ineffective or may require repeated application (Carpenter & Murray, 1999). Any effort to control undesired grasses with herbicides should be accompanied by a long-term monitoring program to determine the relative coverage and boundaries of the target grasses.

5) Revegetation

Native and nonnative species co-occur in grasslands, making for a highly competitive environment. There may be up to several thousand seeds or seedlings within a 10 cm radius. Thus, revegetation projects that rely solely on seeding without reduction of nonnative annual species typically have low success rates (D'Antonio et al., 2002). Fuel treatments combined with native plant revegetation can help to increase the likelihood that native species will be able to reestablish.

Prescribed fire treatments can be used in combination with reseeding during the first year of treatment in areas where native species are not particularly abundant in order to prevent reestablishment by invasive annual grasses in disturbed areas (Klinger et al., 2006). A two to three year combination of burning, herbicide application and reseeding can be used to control sites heavily invaded by cheatgrass (Carpenter & Murray, 1999). When rehabilitating a burn area with native seeds it is advisable to restrict livestock grazing (Paysen et al., 2000).

2. Alkali Meadows

Fire may benefit alkali meadows and dominant saltbush species in certain circumstances.[32] In a study by Pritchett and Manning (2009), who examined the

[32] In general, salt grass (Hauser, 2006) and alkali sacaton have exhibited mixed responses to fire throughout their ranges. The long-lived, subsurface buds of alkali sacaton and salt grass exhibit a degree of resiliency to fire (Pritchett and Manning,

effect of the 2007 Inyo Complex fire in the Owens Valley, rapid recovery of grasses including the dominant species alkali sacaton (*Sporobolus airoides*) and salt grass (*Distichlis spicata*) occurred, while invasive shrub species were absent following the fire. On both burned and control sites, however, groundwater depth was found to play a significant role in determining grass cover, which increased as depth to water decreased. These findings indicate that fire can help to remove shrubs in areas with adequate water supply, but will lead to a shift in community composition toward shrubs where the water table is lower. In addition, Hansen (1986) studied the impact of prescribed fire in alkaline grassland and vernal pool habitat in the Tulare Basin. Relative composition of native species tended to be higher in burned areas, while annuals typically declined after the fires. Native perennials, including *D. spicata*, and native and nonnative legumes and forbs generally had a positive response to fire.

Prescribed burning of alkali meadow habitat, in conjunction with vernal pools and wetlands, has been conducted in the Sacramento National Wildlife Refuge complex for the purpose of reducing invasive species, namely bulrush and cattails, that are thought to suppress vulnerable native vegetation and increase fuel loads. Projects have reportedly decreased fuel loads and enhanced biodiversity (U.S. Fish and Wildlife Service, 2006). Prescribed fire is used in tamarisk systems, of which sacaton and salt grass species are often a part, to control invasive tamarisk and reduce fuel loads. Within these vegetation communities, fire tends to have favorable effects for herbaceous and perennial species at the expense of woody plants (Racher and Britton, 2003). Prescribed fire is also used at the San Luis National Wildlife Refuge to reduce yellow star thistle in alkali sacaton grasslands (Harvey, 2003).

3. Riparian Areas

a. Drivers and Background Fire Regimes

Riparian ecosystems generally have a longer fire frequency and burn less intensely than the surrounding ecosystem. Shaded canopies reduce fire severity and frequency by protecting the understory from wind, resulting in lower evaporation and higher fuel moisture, humidity, and soil moisture. Soil moisture and fuel moisture are further augmented by proximity to the water table, which can significantly decrease fire intensity (Petit and Naiman, 2007). Under certain conditions, however, riparian zones can have more severe fires than surrounding areas. Riparian zones have much higher fuel loads than surrounding areas because of lower fire frequencies and denser vegetation. When extended droughts dry this fuel out, riparian areas become susceptible to fast-moving, high intensity fires, and can even act as corridors spreading these fires through the landscape (Dwire and Kaufman, 2003).

2009). Salt grass may regrow from a rhizomatic root system following fire (Hauser, 2006; Pritchett & Manning, 2009), and seeds can persist through fire (Hauser, 2006).

Susceptibility and intensity of fire within these regions also depends on stream order. Lower stream orders tend to be more susceptible to fire, while higher orders tend to be less susceptible. Because the headwaters of streams are of lower order, and thus more susceptible, fire impacts in these regions will affect the entire watershed downstream through the release of woody debris, ash, and nutrients (Petit and Naiman, 2007). The Ranch contains a multitude of stream orders, seasonalities, and vegetation communities, and thus has multiple riparian fire regimes.

b. Effects of Fire

The effects of fire in riparian systems depend on the intensity of the fire as well as the size and slope of the stream, with first and second order streams experiencing a greater proportion of burn than higher order streams (Minshall et al., 1997). High intensity fires consume water-holding litter, kill plants that would otherwise contribute to transpiration, and reduce the water-holding capacity of soil by destroying its structure (So. Fire Science, n.d.). The combined result—and the single largest effect of fire in riparian areas—is a significant increase in runoff, particularly in areas with steep slopes.

Fire has complex effects on nutrients and water quality, but a general rule of thumb is that high intensity fires decrease nutrient availability while increasing the quantity of nutrients in the water.[33] Because some nutrients are volatilized and others are lost through erosion and leaching, there may be a short term loss of available nutrients (Petit and Naiman, 2007). In the long run, however, fire contributes to cycling nutrients such as carbon and iron (So. Fire Science, n.d.).

Fire also affects water quality, as erosion from increased runoff and streamflow scours and widens the stream, creating major changes in sediment distribution (Petit and Naiman, 2007). The increase in sedimentation negatively affects water quality parameters such as turbidity and dissolved solids. Fire can also create long term temperature changes by reducing canopy cover, thereby increasing biotic activity and depleting dissolved oxygen in the years immediately following a fire (Neary et al., 2005).

The effects of fire on vegetation are mixed. Increased runoff and erosion can make it difficult for riparian vegetation to recolonize, though certain riparian species are fire-adapted. Cottonwoods and oaks, for example, quickly resprout from dormant

[33] But fire can also change certain nutrients inside the biotic system (in the form of dead litter, vegetation, etc.) to a biologically available form (Neary et al., 2005). Often, a fire will result in an immediate increase in nitrogen and phosphorous, which soon return to baseline levels (Neary et al., 2005; Minshall, 1997). Fire increases other nutrients, such as K, Ca, Mg, Cu, Fe, Mn, and Zn, mainly by volatilizing them from vegetation and litter (Neary et al., 2005).

branches after fire (Dwire and Kaufman, 2003). Poison oak grows large, tough root systems that can resprout after fire, and its seeds increase germination after chemical exposure to fire (Howard, 1994).

In addition to its effects on soil, water quality and vegetation, fire impacts aquatic organisms. Macrobenthic communities are affected by many of the same factors as vegetation. High temperatures and ash during the fire can kill off large parts of the population in low ordered streams. Longer term effects such as sedimentation and erosion negatively affect organisms but the overall effect varies significantly between streams. Streams will usually stabilize between seven and ten years after a fire; however, they often still show differences as compared to unburned streams. In some cases, fire causes shifts in the macroinvertebrate community that persist after fire recovery of the ecosystem (Minshall, 2003).

Minshall's 2003 study on Yellowstone postfire response classifies the changes in macroinvertebrates by functional group changes. Large increases were seen in the relative abundance of miners (a collector) for the first three years postfire, after which these species returned to reference level. Scrapers decreased while predators and filterers did not change. An evaluation of the largest taxa revealed that *Chironomids* increased in postfire years one and two, making up over 40% of the assemblage. It appears that *Baetis* and *Chironomids* increased in abundance because they have short generation times, disperse easily, and are opportunistic. Conversely, *Rhithrogena spp*. decreased substantially. Overall, in the first year there was a decrease in abundance, richness, and diversity of macroinvertebrates. This result was attributed mainly to charcoal in the streams. These measures quickly returned to normal. The long term changes were seen in the assemblage rather than measures of community abundance (Minshall et al, 1997).

Periphyton—a mixture of algae, microbes and detritus often found on underwater surfaces—has been shown to decrease after prescribed fires and then rebound within the first year (Beche et al., 2005). In the Yellowstone fire study, however, periphyton increased immediately after the fire, decreased the first year, and then increased again in following summers (Minshall et al, 1997). In theory, an increase in light from a reduction in canopy and increase in nutrients should lead to an increase in periphyton. In some cases, however, postfire sedimentation can bury periphyton and scour the beds, reducing the amount of periphyton. Much like macroinvertebrates, the overall abundance of periphyton can be varied but there is usually a shift in composition postfire. Earl and Blinn's study of the Gila River showed that periphyton shifted to small adnate species (2003). The response of algae to fire is variable depending on the interaction between light availability, nutrients, temperature, and sediment deposition, making a generalized response hard to predict.

c. Changing Fire Regimes

The largest disturbance affecting riparian fire regimes is climate change. Increasing temperature combined with more frequent drought events could lead to lower fuel moisture in riparian areas, thereby making extreme fire conditions more common. The result would be increased fire frequency and intensity in these areas. Moreover, climate change will alter flow regimes dramatically, increasing the dry season and pushing the snowmelt date earlier in most places (Whitlock et al., 2003).

Riparian fire regimes have also been influenced by human land use. In many parts of the Ranch, extensive grazing and water extraction have changed vegetation composition and pushed fire regimes closer to those of surrounding upland areas (Dwire & Kauffman, 2003). Other anthropogenic impacts include water diversions, damming, removal of dead and woody material, soil compaction, and manipulation of channel morphology.

d. Management Approaches

Fuel management affects peak flow, runoff, and water quantity of streams. It can also alter canopy cover, directly affecting which species are able to live in the streams.[34] In general, fuel treatments to the canopy will most strongly affect water quantity while treatment of surface fuels will change sedimentation in the watershed. When the canopy is removed, there is a decrease in both interception and transpiration, resulting in higher amounts of water. A previous study estimated that 15-20% of trees must be removed for this effect to be statistically significant, with the number of trees removed proportional to the increase in water. But in areas where there is little precipitation (less than 18-20 inches), the increase in evaporation from the soil cancels out the decrease in transpiration and interception (Elliot et al., 2010). For this reason, the water supply in Tejon is unlikely to be significantly impacted by fuel management activities.

All fuel management requires the use of roads and vehicles. Roads increase erosion into the watershed, but this effect can be mitigated through decreased tire pressure, better road design, and brushed-in roads. Road use has a greater impact on streams when the roads are hydrologically connected to the streams. The closer or

[34] An important management consideration is the control of invasive species. Giant reed (*Arundo donax*) and tamarisk disrupt native riparian ecosystems and can alter fire regimes. These species are dominant throughout many riparian zones in California and present on the Ranch (Bell, 1997). A variety of methods may be used to control these species. In addition, varieties of grape (*Vitis spp.*) are known to dominate many of the Ranch's riparian zones. While there is no evidence linking fire to increased grape seed germination, it is suspected that grape vines are ladder fuels that carry fire into tree canopies (Howard, 1993).

more connected a road is to the stream, the more runoff will be delivered to the stream (Elliot et al., 2010).

1) Grazing

Grazing may have a disproportionate effect on the Ranch's riparian areas and wetlands, because the most commonly-used cattle breeds tend to congregate around water (Huntsinger et al., 2007). This can have a number of harmful effects, including soil compaction, accelerated erosion and degraded water quality (Appelbaum et al., 2010). It can also disrupt the nitrogen cycle, as cattle consume nitrogen-fixing vegetation and deposit nitrates directly into the water. Moreover, grazing can favor invasive weeds, as cattle not only transport the seeds to riparian areas, but also weaken native species that might otherwise compete with them. On the other hand, grazing has been shown to decrease evapotranspiration rates, which may be beneficial if water-demanding invasive plants are causing streams or pools to dry up before invertebrates can complete their life cycles (Huntsinger et al., 2007; Marty, 2005).

2) Thinning

Seedlings and small tamarisk plants can be removed by hand. If the entire plant is removed, including roots, the plant is unlikely to reestablish. Larger plants can be removed by cutting, but shrub stumps must be treated with herbicides to avoid vigorous resprouting (Lovich, n.d.). Tamarisk control programs should be monitored to ensure that resprouts are controlled and that the plant does not reinvade treated areas.

3) Prescribed Fire

The consequences of prescribed fire in riparian zones are not well researched. Changes resulting from prescribed fire applied to white fir upland zones and incense cedar dominated riparian zones have been found to result in reductions in understory cover and taxa richness. Only a few differences in the in-stream environment, including in periphyton, were briefly detectable, and it was suggested that the riparian zones serve as a "filter" for stream systems after fire (Bêche et al, 2005). It has also been reported that willow growth can be enhanced by prescribed fire, though herbivory after fire may result in setbacks (Dwire et al., 2011).

Despite being largely absent from the literature, prescribed fire is employed in riparian areas by the USFS, Fish and Wildlife Service, BLM, and National Park Service for the purpose of managing fuel (the most widely cited use), as well as restoring riparian zones, supporting wildlife populations, reinstating natural fire cycles, and controlling invasives. A 2010 survey of these agencies revealed that controlled burning is employed more than any other fuel reduction technique, though the majority of riparian management included a multi-method approach. Overall, the survey indicated that burning achieved or partially achieved many management goals, especially fuel reduction. Potential drawbacks, however, include sensitive species restrictions, unknown impacts to stream habitats, cottonwood loss, and fires too cool to adequately reduce fuel and prevent high severity wildfires (Dwire et al., 2011).

Higher severity fires typically cause more erosion than low intensity fires. In high intensity fires, most or all of the ground cover is consumed, leaving soil bare to wind and rain. In extremely high temperature fires, soil can even become hydrophobic, causing major increases in runoff. Prescribed fires can be manipulated to leave material on the ground, thereby protecting the soil from erosion. The amount of duff on the ground and the water content of that duff are the two biggest factors in the amount of duff consumed (Elliot et al., 2010).

4) Herbicides and Revegetation

Herbicides are frequently used to control tamarisk, especially in combination with mechanical treatment. They can be applied to foliage or basal bark of intact tamarisk plants or to stumps following cutting or burning. Large infestations of tamarisk, however, may need to be thinned or burned prior to treatment with herbicides (Lovich, n.d.). Although Carpenter (1998) found that tamarisk can be removed without other invasive species colonizing removal areas, revegetation can be used to increase the likelihood of native species establishment.

4. Wetlands

Wetland fire history has been reconstructed across the United States by examining pollen and charcoal records, which show a history of surprisingly frequent burns. Wetland fires are dependent on seasonality, hydrology and soil moisture. Most occur during the summer, and two studies have shown a correlation between longer hydroperiods and increased fire frequency (Neary et al., 2005). The intensity of fire is determined in part by soil moisture. When soils are wet, there tend to be low intensity fires that consume relatively little organic matter.

Fire has direct effects on wetland vegetation, including the removal of plants and soil organic matter, but it can have indirect effects as well. For example, it affects microorganisms in the soil, thereby changing the rate of decomposition.[35] Fire can also increase soil temperature, which will increase the germination rate of some vegetation. One paper suggests that low intensity fire will lead to bog-type wetlands while high intensity fire will lead to more vascular plants. Other studies have found that low severity surface fires are important in maintaining the herbaceous vegetation

[35] This relationship may also work the other way, as areas with slower decomposition accumulate more fuel, thereby increasing fire frequency (Neary et al., 2005).

of emergent wetlands, and that wetlands with scrub-shrub vegetation generally have more extreme fire behavior than forested wetlands (Neary et al., 2005).

There is a long history of using prescribed fire in wetland areas to reduce litter, promote specific vegetation for wildlife use and create a mixture of habitats for birds (Flores et al., 2011). Controlled burns can also increase plant productivity by allowing more light to reach the surface.[36] One study examined fall burning effects on a northeastern California wet meadow, and found no significant effect on native/nonnative dynamics, but a significant effect on species diversity and individual plant species. The study also found that geese showed a strong preference for the burned plots, suggesting that prescribed burning in the fall can increase forage for birds in the spring (McWilliams et al., 2007).

5. *Oak Woodlands*

a. *Drivers and Background Fire Regimes*

Evidence from ethno-ecological studies suggests that California Indians used fire to manage oak woodlands for over 3,000 years. It is believed that these "prescribed fires" were conducted frequently, with some areas even being burned annually, although the spatial extent of the oak woodlands managed this way is unknown.[37] The effect of this frequent, low intensity burning would have been to reduce understory vegetation such as shrubs and conifers, resulting in open, park-like oak woodlands (Purcell & Stephens, 2005).

b. *Fire Effects and Adaptations*

Many oak species have evolved adaptations that enable them to survive periodic low intensity fires. Some mature oaks are protected from fire by thick bark, which insulates the underlying cambium from the heat of the fire (McCreary & Nader, 2011). Oak seedlings and saplings that are "top-killed" by fire (meaning that the aboveground stems are killed while the root crown survives) have the ability to resprout from the root crown. In addition, oaks have a relatively high resistance to decay, which helps them to recover following fire scarring (Abrams, 1992; Fry, 2008; Pavlik et al., 2002; Standiford & Adams, 1996; Tinnin, 1996).

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[36] Questions remain, however, about whether burned marshes still move organic matter to adjacent areas (Flores et al., 2011)

[37] The Foothill Yokuts were known to manage oak-ponderosa pine forests in order to favor grasses and forbs, affect mushroom production and control fire risk. Oak savannahs and woodlands were managed to promote grass and forb growth, while maintaining acorn production. Fires were also used to open forests, with burning and hand weeding preventing the forest from moving into open prairies and meadows. The reduction in brush limited the severity of natural fires (Anderson, 2006).

Periodic low intensity fires can enhance oak survival and recruitment by reducing the density of understory vegetation, removing successional species, controlling pathogens and invasive species, and mobilizing nutrients (Pavlik et al., 2002; Abrams, 1992). Fire reduces the density of understory conifers and shrubs in oak woodlands, and also temporarily reduces grass and forb cover (Purcell & Stephens, 2005). The reduction in understory vegetation may enhance oak regeneration by lessening competition and increasing the amount of light reaching the soil (Fry, 2002). Low intensity fires can effectively control invasive shrubs and grasses and promote native understory species (Holmes et al., 2011).

Oak woodlands support a rich diversity of wildlife. Vertebrate species richness in oak woodlands may be the highest of any habitat type in California (Tietje & Vreeland, 1997). While fires may temporarily displace wildlife or reduce habitat, low intensity fires may improve wildlife habitat in the medium-term (Vreeland & Tietje, 2001). High intensity fires, on the other hand, can have serious adverse effects on habitat (McCreary, 2004).

Bird species richness has been found to be higher in oak woodlands with high habitat complexity and structural diversity. Fires that increase habitat complexity might thus be expected to increase bird species richness. The actual effect of fire on habitat complexity is variable. For example, fire may increase complexity by creating edge habitat, or it may decrease complexity by removing snags. A study of the relative abundance of breeding birds in mixed blue oak-coast live oak woodland found no change following a low intensity prescribed fire (Purcell & Stephens, 2005).

High intensity fires, which have become more common in oak woodlands as a result of fire suppression and consequent fuel accumulation, have a number of adverse ecological effects. High intensity fires often kill mature oaks, thus reducing habitat for a wide range of species. In addition, high intensity fires can increase rates of soil erosion, which can impair water quality and affect the types of plant communities that a site can support (McCreary, 2004; Standiford & Adams, 1996).

1) Valley Oak Woodlands

Large valley oaks can withstand low intensity ground fires, but will succumb to high intensity crown fires. Valley oak seedlings and saplings are often top-killed by understory fires, but will readily resprout from the root crown (Howard, 1992a). One study found that top-killed saplings sustained a higher growth rate than unburned saplings for two years after a fire. Within 2-3 years of a fire, the height of top-killed saplings was comparable to the height of unburned saplings (Holmes et al., 2011). Valley oak acorns buried underground by animals often survive fire (Howard, 1992a).

In some areas, fire suppression has resulted in the invasion of valley oak woodlands by other tree species, although this is typically a problem in wetter regions and there is no evidence that this is occurring at Tejon Ranch. Live oaks and shrubs commonly encroach on valley oak woodlands at low elevations, while high elevation woodlands are invaded by conifers such as *Pinus ponderosa* and *Pinus coulteri* (Griffin, 1976). The increase in understory vegetation and downed fuel associated with these invasions raises the threat of high severity fire (Howard, 1992a).

2) Blue Oak Woodlands

Blue oaks are fire-tolerant, but wildfire does not appear to be necessary or beneficial for their establishment, growth, or survival (Tyler et al., 2006; Swiecki & Bernhardt, 2002). Blue oak saplings below a certain size are likely to be top-killed by fire. One study found a threshold size of 1.5-2 m below which saplings were top-killed by fire, although this threshold will vary depending on fire intensity and site conditions. Top-killed blue oak saplings often resprout vigorously, but because of the lost height and biomass, saplings damaged by fire take longer to reach the overstory than unburned saplings (Swiecki & Bernhardt, 2002; Bartolome et al., 2002). Saplings damaged by fire are also more susceptible to herbivory. Infrequent fire does not appear to increase blue oak recruitment, and frequent fire may suppress recruitment (Swiecki et al., 1997; Tyler et al., 2006).

Large blue oak trees are less susceptible to fire than small trees due to their thicker bark (Horney et al. 2002). But even large blue oaks may be killed by high intensity crown fires (Wills, 2006). Ladder fuels increase the likelihood of a crown fire occurring, and high tree densities increase the likelihood of the crown fire spreading to nearby trees (Horney et al., 2002).

3) Black Oak Woodlands

Historically, black oak woodlands likely experienced a low severity or mixed severity fire regime. Surface fires occurred frequently in the summer and fall, while moderate to high intensity fires occurred less frequently and may have resulted in occasional stand replacement (Fryer, 2007; Kauffman & Martin, 1986; Van Wagtendonk & Fites-Kaufman, 2006; Keeley, 2006b).

Black oaks have a number of adaptations to periodic fire, including thick bark, a large root system with ample nutrient reserves, and the ability to resprout from the root crown (Tappeiner & McDonald, 1980). Black oak seedlings grow best under full sunlight, and thus black oak recruitment is enhanced by periodic fires that create small gaps in the canopy (Plumb & Gomez, 1983). During long fire-free periods in areas where conifers and black oaks are co-dominant, the shade tolerant conifers can outcompete black oaks, eventually excluding them from a site (Kauffman & Martin 1986; Swiecki & Bernhardt 2002; Fryer, 2007).

High fuel loads inhibit black oak germination and increase the risk of high intensity crown fires (Kauffman & Martin, 1986). Mature black oaks are typically killed by high intensity crown fires, but top-killed trees will readily resprout (Fryer, 2007; Plumb & Gomez, 1983). Many of the black oaks on Tejon Ranch are multi-trunked, suggesting that the trees resprouted following a fire (M. White, personal communication, December 20, 2011).

4) Canyon Live Oak Woodlands

Prior to European colonization, fires in canyon live oak woodlands typically occurred every 35 years or less (Arno, 2000; Paysen et al., 2000). In the Sierra Nevada, canyon live oak-mixed conifer woodlands are reported to have had a pre-1850 median FRI of approximately 11 years (Skinner & Chang, 1996). Fires occurred mainly in the summer and fall and tended to be of low or moderate severity. Fires were less frequent in areas of steep terrain, due to the lack of understory fuel (Tollefson, 2008).

Canyon live oaks are sensitive to fire. Their thin, flaky bark ignites readily and provides poor protection for the underlying cambium (Plumb & Gomez, 1983). In addition, canyon live oaks often form dense canopies, making them susceptible to crown scorch and top-kill from even low severity fires (Skinner et al., 2006). Like other oak species, however, top-killed canyon live oaks rapidly resprout following fire (Tollefson, 2008; Minnich, 1980). As with black oaks, many of the canyon live oaks on Tejon Ranch are multi-trunked, suggesting that the trees resprouted following a fire (M. White, personal communication, December 20, 2011).

c. Changing Fire Regimes

European settlement appears to have initially increased the fire frequency in some oak woodlands. From the late 1800's to the 1950s, ranchers often used fire as a tool to increase the availability of forage for livestock, burning rangelands and oak woodlands every 8 to 15 years (Standiford & Adams, 1996). Beginning in the 1940s and 1950s, however, fires were actively suppressed in many oak woodlands. Fire suppression has resulted in an increase in fuel loads, tree densities and canopy cover, and has enabled conifers and shrubs to encroach into oak woodlands in some areas (McCreary, 2004). The increased fuel load has magnified the risk of large, high severity crown fires, which have a variety of adverse effects on oak woodlands (Fry, 2008; Purcell & Stephens, 2005; Griffin, 1976).

d. Management Approaches

1) Grazing

Grazing has been identified as a possible cause of declining blue oak and valley oak recruitment, though there is conflicting evidence on the issue (Allen-Diaz & Bartolome, 1992; Bartolome et al., 2002; Tyler et al., 2006). But grazing may also benefit oaks, and may affect surface vegetation in other ways. Blue oak savannahs have been described as "islands of enhanced soil quality and fertility," and light to moderate grazing may contribute to this effect by increasing the abundance and availability of nutrients such as phosphorus (Dahlgren et al., 1997). In addition, grazing can promote the growth of caged valley oak seedlings by reducing competition from other (unprotected) vegetation (Tyler et al., 2006). Other studies have found that grazing can help control woody shrubs that compete with oak saplings, although many of these studies were conducted in upper Midwestern oak savannahs (Harrington & Kathol, 2009; Ritchie et al., 1998). Closer to the Ranch, a study of blue oak savannahs in the southern Sierra Nevadas found grazer-specific effects on surface vegetation, with horses promoting grass cover and cattle promoting herbaceous perennial cover (Keeley et al., 2003).

2) Mechanical Thinning

The removal of woody, shade tolerant species in oak stands can help reestablish oak overstory structure, although encroachment from shade tolerant species is not a significant issue on the Ranch. Where encroachment occurs, thinning is best done in combination with some form of follow up control, such as herbicide application or prescribed fire, in order to reduce resprouting from stumps and competition from shade tolerant seedlings (Albrecht & McCarthy, 2006). Albrecht and McCarthy (2006) found that a combination of prescribed burning and thinning increased understory light levels and stimulated oak germination while reducing recruitment by shade tolerant species in central hardwood oak forests. Mechanical or hand removal of weeds around sapling oaks immediately before initiating a prescribed burn can help reduce risk to saplings by lowering surrounding burn temperatures (Holmes et al., 2008).

3) Prescribed Fire

Prescribed burning has been used as a management technique in oak woodlands to decrease the threat of high severity fire, promote the growth of feed for livestock (Fry, 2008), and modify habitat to support wildlife (Lathrop and Osborne, 1991; Paysen and Narog, 1990). Prescribed fire may be employed to suppress nonnative grass species (Holmes et al., 2011), decrease chaparral shrub presence, and enhance species richness (Lathrop and Osborne, 1991). Federal policy also includes controlled burning as a treatment option for mixed conifer forests containing black oak for the purpose of fuel management (Moghaddas et al., 2008). By making nutrients available and eliminating other vegetation that competes for resources, controlled burning has also been suggested as a way to support young oaks (Tietje et al., 2001). However, few studies have examined the impacts of prescribed fire specifically in oak woodlands (Fry, 2008).

Research thus far suggests that oaks are generally resilient to prescribed fire. Blue, black, and valley oaks in mixed oak woodlands have exhibited high survival rates and rapid regrowth after prescribed fire. For blue and valley oaks, this may occur even after full crown scorching (Fry, 2008). Valley oak saplings are also able to survive summer or spring prescribed fire and boast a faster rate of growth subsequent to burning, facilitating recovery (Holmes et al., 2011). Research has also demonstrated that prescribed burning of woodland understory may only kill a small proportion of primarily smaller canyon live oak trees, providing an opportunity for "selective thinning" in canyon live oak woodlands (Paysen and Narog, 1990). Burning after mastication has led to significant mortality of canyon live oaks of all sizes due to increased surface fuel loads, whereas burning alone tends to preserve larger trees (Bradley et al., 2006).

In general, blue oaks appear to tolerate burning. High rates of resprouting have been observed in blue oak saplings following top-kill after low to medium intensity prescribed fire (Tietje et al., 2001). Seedling survival of blue oaks appears to be unaffected by a combination of sheep grazing and fire (Allen-Diaz & Bartolome, 1992). However, Bartolome et al. (2002) found that prescribed fire in dense woodlands failed to increase the height of browsed, shrub-like blue oaks beyond that of unburned trees in the long term, suggesting that fire does not aid in blue oak recovery. In fact, excluding both herbivory and fire resulted in greater growth than burning or browsing alone (Bartolome, 2002). Thus, it is possible that prescribed fire has variable effects on blue oaks depending upon whether or not browsing has occurred prior to treatment.

Black oaks seem to be more sensitive to prescribed burning in certain cases. Black oaks can maintain their relative numbers after thinning, prescribed fire, or both in Sierran mixed conifer systems (Stephens and Moghaddas, 2005), but heat-induced damage to more than 20% of the crown during a low intensity fire can inhibit subsequent sprouting in mixed oak woodlands (Fry, 2008). Furthermore, burning may pose a threat to the production of new oaks, as prescribed fire has been shown to reduce the number of black oak seedlings present within a 4 year period of fire, though a similar impact on black oak seedlings has not been observed after a combined thinning and fire approach (Moghaddas et al., 2008).

Burning after the summer dry season may help to ensure that young oaks have the opportunity to successfully rebound (Lathrop and Osborne, 1991). However, spring burning may help to prevent negative effects of sapling top-kill, which may be further mitigated by using prescribed fire only in areas where trees exceed 300 cm and reducing fuel loads in the understory prior to fire (Holmes et al., 2011). Prescribed burns should not be used too frequently, especially at intervals less than 10 to 14 years, as even low intensity prescribed burns can delay the maturation of saplings (Fry, 2008). In addition, prescribed burning should not be used during drought years, as dry conditions increase the risk of uncontrollable wildfire and may inhibit postfire recovery (Howard, 1992a; McCreary, 2004).

4) Herbicides

Herbicides may be used to target invasive annual grass understories or woody, shade tolerant species in oak woodland systems. Nonselective foliar herbicide has been used to eliminate annual grasses around oak seedlings (Bernhardt & Swiecki, 2001). However, seedlings should be shielded from the spray when nonselective herbicide is used. Some broadleaf herbicides can negatively affect oak root growth and are not recommended in oak woodlands (DiTomaso et al., 1997). Herbicides can also be used following mechanical removal in oak woodland systems to reduce sprouting from stumps of trees or bushes which have been removed to reestablish oak overstory cover (Brudvig & Asbjornsen, 2007).

6. Mixed Conifer Forests

a. Drivers and Background Fire Regime

The conifer forests of Tejon Ranch are typically dominated by shade tolerant white fir (*Abies concolor*) and incense cedar (*Calocedrus decurrens*), and shade intolerant ponderosa pine (*Pinus ponderosa*) and Jeffrey pine (*Pinus jeffreyi*), which are collectively known as yellow pines (Appelbaum et al., 2010). Historically, frequent ground fires cleared these areas of woody fuel, brush, and saplings but left mature trees relatively unharmed. Where localized crown fires did occur, they created canopy openings that allowed shade intolerant species to regenerate on the mineral soil exposed by fire. This mosaic of age classes created a highly variable and healthy forest structure with natural "fuel breaks" that limited the extent and severity of wildfires (Verner et al., 1992).

Most evidence suggests that presettlement conifer systems in California had FRIs of between 1 and 30 years (Habeck, 1992). The mountainous terrain where conifer systems are generally found influences wind patterns, fuel type and fuel moisture, and can also act as a natural barrier to fire spread. Species composition within these forests is influenced by precipitation gradients, with ponderosa pine, white fir, and incense cedar occurring on moist windward slopes, and Jeffrey pine more prevalent on leeward slopes (Barbour et al., 2007). More precipitation falls on windward slopes while leeward slopes typically have drier fuel loads that are more prone to ignition. Elevation is also a factor, as fires tend to occur less at higher altitudes due to decreased ignitions and lower fuel loads. Fuel production is higher at low elevation areas, and fuels dry out more quickly at low sites, creating greater risk of high severity fires (Taylor, 2000). A study by Beaty and Taylor (2001) found that the median FRI in conifer forests located on more mesic, high elevation, north-facing slopes was 34 years, compared to a FRI of only 9 years on drier south-facing slopes. This suggests that forest density and fuel loading can widely vary based on small-scale geographical contexts.

b. Effects of Fire

Many of the Ranch's dominant conifers are fire-adapted. Yellow pines have thick bark and high crowns when mature. They are also able to self-prune their lower branches to rid themselves of ladder fuels. These traits help protect mature trees from crown fires (Keeley, 2006b). Yellow pine saplings (dbh < 5cm) often succumb to low intensity surface fires, but this helps to maintain lower forest densities (Belsky & Blumenthal, 1997). Yellow pines also have large, well-protected buds, and can lose up to half of these buds before being top-killed. Furthermore, the seeds of ponderosa pine are heat tolerant and can germinate even after exposure to temperatures of 930°C (Wagtendonk and Fites-Kaufman, 2006). Fire is not essential to yellow pine regeneration, but it aids germination by creating sunlit gaps and exposing mineral soil (Habeck, 1992). In the wake of a fire, shrubs generally resprout while herbs and grasses reseed and grow rapidly. Within a few years, however, fir and yellow pines dominate the gap[38] (Wagtendonk and Fites-Kaufman, 2006).

A recent study examining the 2003 fires in Cuyamaca State Park suggests a risk of type conversion after catastrophic fires. In this case, years of fuel buildup caused by fire suppression led to fires that killed many of the mature trees in the mixed conifer forest. The fires also killed conifer seeds, and the distance of burned areas from other conifer forests hindered recruitment of new seeds. Rather than new conifers, various oaks and shrubs sprouted, and soon came to dominate the system (Goforth & Minnich, 2008).

c. Changing Fire Regimes

Fire regimes began to change with the introduction of European settlers into California. Extensive grazing and mining operations altered ignition patterns and vegetative structures. The most dramatic shift occurred in 1905 when the USFS began a policy of total fire suppression. An analysis of fire regimes by Alan Taylor found that FRIs in some conifer systems are now nearly 90 years (Taylor, 2000). Fire prevention in mixed conifer forests has led to decades of stand densification and

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[38] It is believed that the ideal conditions for yellow pine regeneration are a large seed crop, reduced competition from herbaceous species, and abundant rainfall in the spring following germination (Belsky & Blumenthal, 1997).

excessive fuel accumulation, creating the potential for catastrophic crown fires that can kill very large tracts of forest (Verner et al., 1992). Fire suppression has also shifted species composition toward more shade tolerant and fire-sensitive species, such as white fir and incense cedar[39] (Taylor and Skinner, 2003). In addition, it has resulted in the shading out of many shrubs such as deer brush and mountain misery (Wagtendonk and Fites-Kaufman, 2006).

As fire suppression creates denser forests, conifer systems face additional water stress during dry seasons and drought. Competition for limited water resources by an excessive number of saplings and pole-sized trees can harm fire-resistant mature trees and reduce the resilience of the forest as a whole (Goforth & Minnich, 2008). As a result, growth can be suppressed and weaker trees can become vulnerable to insect and disease attack. Large tree mortality can create additional fuel (Belsky & Blumenthal, 1997).

In addition to the legacies of fire suppression, conifers face a number of adverse effects from climate change. Evapotranspiration rates are expected to increase, which will reduce moisture availability in the soil and increase the onset of drought conditions. This, in turn, will intensify water stress and reduce the suitable range of conifers. Problems associated with tree stress from overcrowding will further strain forests by exacerbating disease and insect attacks. Additionally, climate change is likely to dry out fuels sooner, which will increase fire risk by extending the fire season. These threats serve as added incentive to maintain forests at lower densities and reduce fuel loads (Battles et al., 2008).

d. Management Approaches

1) Grazing

Grazing in conifer stands negatively impacts forest health. Studies comparing grazed and ungrazed plots protected by exclosures have found that grazed plots had nearly twice as many tree seedlings. Livestock browse down the understory layer of grasses and shrubs, reducing the competition faced by conifer seedlings and enabling them to recruit in very large numbers (Belsky & Blumenthal, 1997; Miller & Urban, 2000). This effectively replaces surface fuels with ladder fuels, causing the risk of low intensity surface fires to decline while the risk of high intensity crown fires increases (Belsky & Blumenthal, 1997). Grazing also tends to reduce soil moisture and increase runoff as surface soils are compacted by animal trampling and litter is removed. This reduces water availability, which can be an additional stressor for

[39] Despite the fact that fire regimes in mixed conifer forests have been drastically altered, invasive species have had limited impact on these areas. The majority of invaders are herbaceous plants, and only a few nonnative woody species are appearing, such as scotch broom, tamarisk, Russian olive, and tree-of-heaven.

conifers (Appelbaum et al., 2010). In drought or overcrowding conditions, the result can be increased mortality and higher fire risk (Belsky & Blumenthal, 1997).

Grazing also favors invasive species by creating fuel breaks, which act as corridors into forests and shelter invasive seeds during fire. Cattle can amplify this effect by transporting additional invasive seeds along the corridors, with particularly noticeable results when grazing is conducted after a prescribed burn (Keeley, 2005; Keeley et al., 2003). For this reason, if grazing is to be used in these communities at all, Keeley (2003) recommends that it be decoupled with prescribed fire.

2) Mechanical Thinning

Mechanical thinning treatments aimed at reducing forest density and fuel loads can improve the health of conifer systems. The common paradigm in conifer management involves removal of smaller age cohorts that increase forest density and ladder fuels, and retention of mature, fire resistant trees. When a forest is thinned, the canopy is opened, which allows more light to hit the forest floor. This can result in drier surface fuels but wetter soils, because the canopy intercepts less precipitation. Wetter soils, in turn, provide more water for mature trees, thereby promoting tree health (Ma et al., 2010).

The tree species and size classes selected for thinning can change forest composition and impact phosphorous and nitrogen availability. A study conducted by Miesel et al. examined the effects of two different thinning treatments on nutrient levels in forest soils. One treatment was designed to favor ponderosa pine while the alternative treatment favored retention of large mature trees regardless of species. The former treatment was aimed at achieving 80% forest composition of ponderosa pine by basal area, followed by retaining sugar pine, all white fir >76 cm dbh, and all incense cedar >25 cm dbh.[40] The latter simply retained all trees > 76 cm dbh and thinned the remaining smaller vegetation. The results indicated that soils experience a short-term nutrient increase following the size-preference treatment. In the long term, however, the higher quality organic matter from ponderosa pines in the pine preference treatment contributed greater nutrient inputs to soils. This suggests that thinning with an emphasis on retaining ponderosa pine can lead to improved forest health and ecological functioning (Miesel et al., 2008).

Studies in pure white fir stands have shown that thinning trees less than 9" dbh and 11 feet tall can significantly reduce fire risk (Zouhar, 2001). In the absence of a wildland fire policy, treatments need to be occasionally reimplemented in order to reduce new growth. Monitoring fuel loads and densities is important in

[40] Most literature favors the preservation of shade intolerant yellow pines at the expense of shade tolerant white fir and incense cedar, and advocates the removal of trees less than 25 cm dbh (Schmidt et al., 2008).

determining when to re-thin, and it may also be reasonable to conduct thinning on a time frame similar to the presettlement FRI (Verner et al., 1992). Additionally, as mechanical thinning can be quite expensive, one strategy for minimizing the area requiring treatment is to utilize the Strategically Placed Area Treatments (SPLAT) approach. By thinning particularly fire prone areas such as south facing slopes and ridgetops, managers can essentially create "speed bumps" that can slow or contain severe wildfires. The thinning of particular treatment areas can also help to create the mosaic of age structures that is associated with healthy forests (Schmidt et al., 2008).

Potential drawbacks of mechanical thinning include the wounding of up to 50% of residual trees, and increases in surface fuels created by material left on the forest floor after thinning (Zouhar, 2001). New surface fuels can dry out quickly as sunlight reaches the forest floor. One strategy for addressing this is to masticate smaller trees and snags (2-25cm dbh) and process downed wood material in a chipper (Stephens, 2005). While masticated material poses less of a fire hazard, a study by Miller et al. found that the processing of woodchips onto the forest floor is associated with lower species richness, diversity, and overall plant cover than unchipped plots (Miller et al., 2009). A more common approach is to follow thinning operations with a prescribed ground fire to clear away the surface fuels.

3) Prescribed Fire

Prescribed burning has been applied in Sierran mixed conifer forests to reduce fuel loads resulting from fire suppression and prevent high severity fires (Vallaint et al., 2009). More specifically, controlled burning is employed to remove surface fuels and small trees (van Mantgem et al., 2011). Prescribed fire is also used to restructure forests to approximate historical forest conditions (North et al., 2007).

Studies suggest that prescribed burning is effective at altering fuel loads and influencing forest fire "resiliency," defined by Agee and Skinner (2005) as "maintaining substantial live basal area after being burned by a wildfire." Fuel modifications that encourage resiliency include minimizing surface fuels to prevent the movement of fire into the canopy, extending the distance between flame reach and live canopy, and preserving large trees that have the ability to withstand fire. The first two objectives may be met by prescribed fire, but fires would have to be of excessive severity to achieve density goals. Crown fire is further suppressed by decreasing the concentration of canopy fuels (Agee and Skinner, 2005).

The ability of fire to restore historic conditions is uncertain and may depend upon fire intensity. High intensity prescribed fires have been used to create new canopy openings, encourage tree clumping, and reduce shade tolerant understory species (Schmidt et al., 2006). Schmidt et al. (2006) found that low intensity burns effectively reduce fuel loads but do not lead to overall restoration of forest structural components. Van Mantgem et al. (2011) posit that the reductions in density of small trees observed in Sequoia National Park aided in forest restoration, though significant adjustments to species composition were not observed. North et al. (2007) found that a combination of thinning and burning led to proportions of shade tolerant and intolerant species similar to those indicated by a reconstruction of forest conditions in 1865. In contrast, Valliant et al. (2009) determined that prescription burning had little effect on structure in general, but did observe a decline in surface and ladder fuels.

Studies have highlighted several drawbacks to the use of prescription burning in Sierran mixed conifer forests. Treating small sites may be insufficient to decrease fuel continuity and prevent the spread of some fires (Agee and Skinner, 2005). Furthermore, because trees in sites treated with prescribed fire are more likely to survive the heat of a subsequent fire the further they are from the edge of the treated plot, managing very small sites (0.5 ha) may not be sufficient to prevent tree mortality (Ritchie et al., 2007). An additional negative consequence of prescribed burning is the possibility of increased influx of invasive species[41] (Keeley, 2006a; Knapp et al., 2007). However, if forests are not managed, crown fires resulting from fuel buildup have the potential to create large canopy gaps that favor invasive species colonization (Keeley, 2006a). To address this dilemma, Keeley (2006a) suggests removing grazers from the area prior to burning to prevent the introduction of invasive species.

Other issues include the difficulty of applying a particular fire prescription and the uncertainty surrounding the outcome of a fire (Schmidt et al., 2006). A single burn may not be adequate to achieve or maintain management objectives. Repeat burning may be advisable to clear away fallen trees killed by the initial prescribed fire (Mutch and Parsons, 1998; Vallaint et al., 2009). Van Mantgem et al. (2011) and North et al. (2007) also suggest that more than one burn may be needed to achieve target density levels, even though in the case of the van Mantgem et al. (2011) study, fire was of higher severity and considered largely successful.

4) Herbicides

Shrubs that take advantage of increased light can establish after fire in conifer systems. These shrubs may inhibit conifer seedling growth and increase competition for water, but can sometimes be controlled with herbicides. Broad spectrum herbicides have been shown to enhance native plant diversity when applied after catastrophic fires by controlling shrubs and forbs that compete with conifers (DiTomaso et al., 1997). However, there is also evidence that herbicide treatments to

[41] Research by Keeley et al. found that unburned conifer stands generally had very few invasives, while burned stands had a significant number of invaders (Keeley, 2003). These studies indicate that nonnative species tend to increase after fire, which can potentially alter ecosystem functions and compromise fire hazard reduction in the long-term (Kane et al., 2010).

remove shrubs have resulted in enhanced grass fuel loads that are sufficient to carry fires (Keeley et al., 2011).

7. *White Fir Stands*

a. *Drivers and Background Fire Regimes*

The Ranch supports several large tracts of natural white fir stands. Determining best management practices for these areas is difficult because much of the literature dealing with mixed conifer forests treats white fir as a pest species that should be thinned from the system. The shade tolerant white fir is susceptible to high severity fires due to fairly low crown heights and branch retention, along with high stand density that can act to carry a wildfire. Despite the associated fire risk, natural white fir stands provide important habitat for many species, including California spotted owl, deer, elk, and bear (Zouhar, 2001). Maintaining the vitality of these white fir stands is an important goal on Tejon Ranch.

White fir seedlings typically germinate most successfully on bare mineral soils created by fires, but seeds can also readily grow on unburned soils covered with an organic litter layer. Survival is aided by the shady conditions provided under canopy cover where seedlings can easily establish.[42] Saplings can persist for many years under low light conditions but growth increases greatly when canopy openings are created, allowing more light to reach the saplings (Zouhar, 2001).

White fir stands may exhibit mixed severity fire regimes or high severity fire regimes (Zouhar, 2001). In northern California, stand age structures suggest surface fires in some areas and more severe fire in others (Stuart and Salazar, 2000). More severe fire may be common in forests composed of white fir and Jeffrey pine in the Mojave region (Zouhar, 2001). White fir is considered to be "fire tolerant" in mesic environments, where fire does not diminish white fir prevalence, and to be "fire-sensitive" compared to pine species in arid environments (Keeley et al., 2009). Historical median FRIs for white-fir-dominated forests along the northern coast of California have been estimated to be between 12 and 161 years with a median of 27 years (Stuart and Salazar, 2000). Kilgore and Taylor (1979) calculated a mean FRI prior to 1875 for white fir, incense cedar, and sugar pine forests of 14 to 17 years.

b. *Effects of Fire*

Mature white firs are moderately fire resistant, as they develop thicker bark with age. However, they remain susceptible due to their tendency to retain low

[42] If there is a nearby seed source and fire-free interval, white fir seedlings can establish under a shrub layer within 10 to 20 years, and eventually shift the species composition to a dominant white fir forest (Zouhar, 2001).

branches with extensive lichen growth, and because they have fairly shallow roots which can be damaged by fire. This is particularly true in stands with deep litter layers where long-smoldering duff on the forest floor can kill shallow white fir roots (Zouhar, 2001). Wildfires can also kill white firs through crown scorching, the loss of fine roots (Mutch and Parsons, 1998), or "cambial heating" which damages sensitive stem areas. Additionally, while a fire may not directly kill a tree, fire-damaged white firs are highly susceptible to insect and disease invasion because fire scars provide a point of entry for foreign materials (Zouhar, 2001).

Ignitable primarily in the dry season, white fir and incense cedar stands are less readily burned than other mixed conifer forests, which contributes to fuel buildup (Agee et al., 1978). Some resilience to crown fire is indicated by observations of epicormic branching from white fir trunks after wildfire, a response that is more pronounced in trees experiencing greater crown damage. Bigger trees also tend to exhibit the most branching, providing some evidence that these trees are able to withstand severe fires that clear the understory of younger trees (Hanson & North, 2006). Furthermore, high severity fires occur naturally (in addition to low and moderate severity fires) within white fir forests with varying composition in Lassen National Forest. Such differences in burn severity are associated with topographic differences (Beaty & Taylor, 2001).

c. Changing Fire Regimes

Fire suppression policies over the past century have facilitated an increase in younger white fir density as saplings have been unchecked by periodic groundfires (Hanson & North, 2006; Parsons & DeBenedetti, 1979). This buildup of fuel in white fir stands has made crown fires more likely. In the southern Sierra Nevadas, forests comprised of white fir only have remained white fir stands under the influence of fire restriction, while white fir and incense cedar stands that contain some black oak and sugar pine have also experienced increases in the proportion of white fir (Parsons & DeBenedetti, 1979). In addition, Parsons and DeBenedetti (1979) determined that composition of a forest dominated by ponderosa pine and black oak changed from 4% to 44% white fir during the period of fire suppression.

d. Management Approaches

It has been suggested that thinning of white fir less than 9 inches dbh and 11 feet tall could help minimize damage from future wildfires (Zouhar, 2001). In overly dense white fir stands, reducing the basal area density to levels between 16-37 square meters per hectare was found to significantly improve tree vigor and health. Further, in white fir stands experiencing more than a century of fire suppression, such thinning operations were found to reduce dry stem fuels by between 41-50 Mg per hectare, significantly reducing fire hazard risk (Zhang et al., 2007).

Fire is used in white-fir-dominated systems for the purpose of fuel management, and prescribed burning has been demonstrated to reduce fuel accumulations by an average of 85%. Repeat burning is likely needed to maintain lowered fuel levels, as the majority of surface fuels can regenerate within 10 years subsequent to burning. However, white fir mixed conifer systems that have not burned for 40 to 90 years may not continue to amass surface fuel due to equivalent rates of fuel buildup and decay, suggesting that burning may not be immediately necessary to impede fuel buildup (Kiefer et al., 2006).

White fir forest structure may also be significantly altered by prescribed burning, as fire can remove large proportions of smaller trees. In white fir forests with limited black oak, incense cedar, and Jeffrey pine presence, prescribed fire has been observed to remove 75% of trees with a dbh of less than 50 cm and about 5% to 6% of tree basal area (Mutch and Parsons, 1998). In another study, fire removed over 60% of trees smaller than 30 cm in diameter (Kiefer et al., 2006).

It has been argued that fire should not be applied to reduce density if white fir maintenance is the ultimate management goal (Zouhar, 2001). Prescribed fire, especially of higher intensity, may be more damaging to white fir and incense cedar than other conifer species such as yellow pine, and was observed to shift composition toward ponderosa pine in Grand Canyon National Forest (Fulé et al., 2004). During prescribed burning in the Blodgett Forest Research Station in California, white fir and incense cedar—the species that comprised most of the understory—experienced higher mortality than other species (Schmidt et al., 2006).

But other studies have indicated that white fir and incense cedar are not severely or disproportionately impacted by prescribed fire. The percentage of ponderosa pine and white fir trees killed by prescribed burning in the Sierras has been observed to be comparable (Mutch and Parsons, 1998), and controlled burning in Sequoia National Forest retained white fir as the dominant species (van Mantgem et al., 2011). One study demonstrated that thinning alone, as well as thinning combined with prescribed fire, failed to diminish the proportion of the dominant white fir. Burning and thinning together actually increased the amount of incense cedar in Teakettle Experimental Forest in Sierra National Park (North et al., 2007). Furthermore, prescribed fire does not necessarily have a negative impact on white fir or incense cedar seedlings, but also may not favor the emergence of new white fir seedlings (Mogghaddas et al., 2008).

Though overall prescribed fire did not generate significant detrimental effects to forbs, graminoids, annuals/biennials, and perennials in the understory of white fir-dominated mixed conifer forest (containing incense cedar) in Sequoia National Park, research has pointed to burns in June as leading to less pronounced species reductions than burning in fall. The observed effects, however, were gone within 2 to 3 years. Given that fall burning has been shown to reduce fuels to a greater extent, fuel-driven fire behavior likely influences understory species more than burn season in areas where fuels have been allowed to accumulate (Knapp et al., 2007).

Tree-centered spot firing has been used in conjunction with strip-head firing in the presence of thinning slash to produce shorter flame lengths near trees that may be otherwise harmed by crown or trunk scorching. Observations indicate that tree-centered spot firing leads to less flame activity near trees as well as fewer incidences of bole and crown ignition when compared to some prescribed burning methods (Weatherspoon et al., 1989). These findings suggest that mechanically treated fuels in white fir stands on Tejon Ranch could potentially be removed by spot firing and prescribed burning so that resulting fuel does not promote potentially severe wildfire.

8. Chaparral

a. Drivers and Background Fire Regimes

Chaparral is comprised of diverse assemblages of drought-resistant and fire hardy shrubs with woody stems and small evergreen leaves that are coated with a waxy layer to reduce moisture loss and overheating (California Academy of Sciences, 2005). Chaparral typically benefits from stand-replacing crown fires at intervals of 20 to 100 years (Conard & Weise, 1998). Short-lived, herbaceous plants colonize burn sites, taking advantage of an open canopy and decreased competition for resources. Shrubs regenerate quickly, producing closed canopies after a period of ten or more years (California Academy of Sciences, 2005).

In southern California chaparral, fires generally fall into one of two categories. They either burn for a period of weeks or months during hot summer conditions in steep terrain, or in autumn coinciding with short periods of extreme fire weather characterized by foehn winds (Stephens & Sugihara, 2006). Foehn winds can exceed 100 km per hour and usher in high temperatures and low humidity in southern California, creating particularly extreme fire weather conditions at a time when natural fuels are the driest (Keeley, 2006b). Wildfires in chaparral are naturally ignited by lightening between mid-July and September (Quinn & Keeley, 2006). Human caused ignitions account for an increasing proportion of chaparral wildfires. Today, lightening ignitions in coastal and southern California shrublands are not good indicators of fire frequency, as they may account for anywhere from 1% to 50% of wildfires per decade (Keeley & Fotheringham, 2003).

Spatial and temporal patterns of ignitions, fuels, weather, and topography all contribute to fire size and severity (Keeley & Fotheringham, 2003; Moritz, 2003). Under extreme weather conditions, the probability of large fires occurring in chaparral is poorly predicted by fuel levels and antecedent climate (Moritz, 1997). The correlation between rainfall patterns and wildfires is relatively weak, as more rain results in both increased fuel and higher fuel moisture. However, there is a negative correlation between autumn rains and fire, suggesting that an early rainfall can cut short the fire season (Keeley & Fotheringham, 2003). The severity and impact of burns may also vary on a fine scale as a function of floristic differences (Keeley, 2006b).

b. Fire Effects and Adaptations

Many chaparral plant species depend on fire for germination and recruitment. Sprouting species regenerate after a fire by forming new shoots from underground root systems or burls. Some sprouting species depend on this mechanism alone for regeneration after fire, while others rely on a combination of seed germination and sprouting. Burls can be hundreds of years old and survive numerous fire events (Quinn & Keeley, 2006). In the prolonged absence of fire, mesic areas may become dominated by sprouting species at the expense of seeding species (Keeley, 1992a). Sprouting species are capable of continuously regenerating the canopy, producing an age structure that is typically less homogenous than canopies dominated by seeding species (Keeley, 1992b).

Chaparral contains a large portion of obligate seeding species, including species of *Arctotaphylos* and *Ceanothus,* which lack any ability to sprout vegetatively after a fire (Keeley, 2006b). These species are typically top-killed by fire and rely on cues from heat, smoke, and charred wood for germination. Obligate seeding species produce the majority of their growth and have the greatest potential to increase their population in the years immediately following a fire (Keeley, 1992b). They may have a competitive advantage over sprouting species following fire after long fire-free periods. The accumulation of dead plant material in the prolonged absence of fire can lead to high intensity fires which kill more sprouting species and provide more openings for seedling establishment (Keeley & Zedler, 1978).

FRIs of less than 20 years can threaten native chaparral vegetation. Short FRIs may limit recruitment opportunities for obligate resprouting species if fire prevents the development of necessary germination microhabitats, although resprouting species are typically more tolerant of short fire intervals than obligate seeding species (Keeley, 1992b). Obligate seeding species will not produce a viable seedbank if fire prevents plants from reaching maturity. Small young shrubs can produce some seeds, but not in the quantity required for stand regeneration. When fires occur at intervals short enough to inhibit species development and critical levels of seed production, obligate seeding species are at serious risk of extirpation (Keeler-Wolf, 2010; Zedler, 1995). Studies in the Santa Monica Mountains have shown that repeated short fire intervals of approximately 12 years cause reductions in shrub densities and a loss of obligate seeding shrubs. Furthermore, short fire intervals can prevent the shrub canopy layer from closing, making burn sites more susceptible to invasion. When invasives colonize burn sites and out-compete native chaparral species, chaparral ecosystems are prone to type conversion. Invasive annual grasses exacerbate fire management issues because they are more likely to ignite, and tend to dry out earlier in the spring, thus extending the fire season (Lambert et al., 2010).

Stands that have not burned for over 50 years have been associated with a higher proportion of dead wood, limited growth, and no new seedling development (Hanes, 1971). Yet studies have shown that as mature stands of chaparral shrubs age in the prolonged absence of fire, they show no signs of dying out or replacement by succession (Keeley, 1992a; Keeley, 1992b). The greatest risk may be to obligate seeding species that rely on fire to stimulate dormant seedbanks, and which may succumb to competition from sprouting species (Keeley, 1992a).

c. Changing Fire Regimes

Modern fire suppression has kept FRIs for chaparral within their natural range of variability (Keeley & Davis, 2007). Studies of pre-suppression era chaparral fire regimes suggest that lightening ignited fires were frequent and burned a relatively small portion of chaparral landscapes while large-scale fires occurred once or twice a century (Keeley, 2006b). Today, fire suppression may contribute to the accumulation of fuel in extensive, contiguous areas (Minnich, 1983); however this has not been shown to significantly increase risk of large scale fires (Keeley & Fotheringham, 2001). Large fire events have not increased in frequency; fueled by extreme weather conditions, they burn through chaparral without regard to the age of vegetation (Moritz, 2003). Younger age classes can serve as natural fire breaks under moderate weather conditions, but have not been shown to limit fire spread under severe fire weather conditions (Conard & Weise, 1998).

A combination of anthropogenic ignitions, habitat fragmentation, and increased suppression since 1950 has led to more small fires in chaparral and an overall decrease in fire size (Moritz, 2003). There is insufficient evidence that fire intensity has increased since suppression began. Seasonality has not changed significantly from natural conditions. Ignitions peak in the summer, but the greatest area burned occurs in the fall under extreme weather conditions when fuels are the driest (Keeley & Fotheringham, 2003). Most large fires occur between September and November (Quinn & Keeley, 2006). Size of fire does not usually influence vegetation recovery because most species recover endogenously (Keeley, 2010).

Where there are high numbers of anthropogenic ignitions, the fate of chaparral is arguably more closely linked with future patterns of human demography than a changing climate (Keeley, 2006b). But there is significant uncertainty regarding how the frequency and intensity of extreme fire weather events will shift with changing climate patterns in the future (Moritz & Stephens, 2006). Global climate models used to simulate changes to chaparral fire regimes do not account for some of the most significant regional factors influencing fire behavior in chaparral landscapes; local precipitation patterns and the frequency of foehn winds are not well represented.

Moreover, increasing atmospheric CO_2 not only influences climate, but may also change plant physiology (Davis & Michaelson, 1995). Despite these uncertainties, studies by Davis and Michaelson (1995) showed that moderate increases in temperature would likely increase total area burned and decrease time between fires in chaparral vegetation. Limited seed dispersal typical of most chaparral species may present challenges to climate change adaptation (Keeley & Davis, 2007).

d. Management Approaches

1) Grazing

Chaparral vegetation is generally more palatable to sheep and goats than to cattle (Narvaez et al., 2011). Goats are often used to create firebreaks in chaparral, and they can significantly reduce the prevalence of Himalayan blackberry (*Rhubus discolor*), poison oak (*Toxicodendron diversilobum*), and other invasive or undesirable species (Huntsinger et al., 2007). In addition to its effects on invasives, grazing can reduce horizontal fuel continuity by limiting the establishment of herbaceous vegetation (Swank & Oechel, 1991).

But grazing can also amplify the effects of fire. A study of chamise (*Adenostomata fasciculatum*) resprouting after fire found that postfire grazing intensity increased with fire intensity, and that both delayed resprouting times (Moreno & Oechel, 1991). To the extent that chaparral species face competition from invasive annual grasses that benefit from fire, immediate postfire grazing may favor these grasses at the expense of native species.

2) Prescribed Fire

Prescribed burning is used in California's chaparral ecosystems for the purpose of wildfire mitigation (Keeley, 2006a). There is some indication that prescribed fire may have some advantages over mastication as a method of fuel manipulation, but not in all cases.[43] Evaluating different methods of fuel reduction, Potts et al. (2010) found that, compared to mastication, prescribed burning resulted in more shrubs within a 3-year period, possibly as a result of fire-generated increases in nutrients. Mastication may retard the regrowth of fuel, but this technique also leaves behind dead fuel, can permit grasses to move in temporarily (Potts et al., 2010), and has been associated with greater invasive species richness and nonnative grass cover. Nonnative grasses would eventually be overcome by shrubs, but fire risk and thus risk of type conversion may be temporarily augmented. However, mastication has also been shown to maintain more native species richness than burning (Potts & Stephens, 2009).

[43] Managing burns safely is a significant challenge, a possible solution to which is to refrain from using prescribed fire between summer and fall (Keeley, 2006a).

Seasonality is likely to play a role in the impact of fuel management. More shrub seedlings have been produced as a result of fuel manipulations in the fall or winter, with prescribed fire during the fall leading to the highest densities of *Ceanothus*, a species that supports black-tailed deer populations (Potts et al., 2010). However, burning in winter or spring has been recommended because of potentially reduced nonnative species presence relative to burning in the fall (Potts & Stephens, 2009), though Keeley (2006a) reported an instance of significant nonnative species proliferation after winter prescribed fire. Similar seasonal effects have not been observed for resprouting (Potts et al., 2010). The appropriate season for fuel treatments may therefore depend on management goals and site conditions.

As some wildfires have been less severe and have failed to move through areas that contain younger chaparral, prescribed fire has been encouraged as a method for removing older chaparral and generating "mosaics" of fuel across the landscape (Dougherty & Riggan, 1982). However, large-scale fires have travelled through areas previously subject to burning, especially when conditions have been conducive to fire (Dougherty & Riggan, 1982; Dunn, 1989). Fires are often fought along the wildland-urban interface (WUI) even where areas of younger chaparral exist because fires spread too rapidly through chaparral to utilize these patches (Keeley et al., 2009). Furthermore, creating chaparral landscapes less than five years old (the age-class reportedly successful at preventing fire spread when winds are high) may result in problems associated with erosion and habitat homogenization (Conard and Weise, 1998). Instead of applying prescribed burns in a mosaic fashion, Conard and Weise (1998) advocate for "strategically placed dynamic fuel management zones" near roads, which would serve to restrict fire movement and would provide areas where suppression efforts could be conducted to control fire frequency and seasonality.

3) Revegetation

Until recently, reseeding following a catastrophic fire in chaparral was a common practice. This was done in order to stabilize the site, control erosion and protect watershed processes. Revegetation typically occurred with fast-growing grasses with little regard to the invasibility of the species. Today, the use of broadcast seeding after fire in California chaparral is broadly discouraged. Very few studies found seeding to actually reduce erosion in chaparral in the first or second year after a fire (Thode et al., 2006). Furthermore, seeding with nonnative species can compete with native vegetation. More commonly, mulch and hay bales will be used today to stabilize banks and limit erosion in chaparral following fire (Keeley, 2006b), although these stabilization methods are of less relevance in wildland areas.

9. *Joshua Tree Woodlands*

a. *Drivers and Background Fire Regimes*

Situated at the western edge of the Mojave Desert, the Ranch hosts at least 2,000 acres of Joshua tree woodlands. Unlike Joshua tree stands in the southern Mojave, which are likely to shrink or disappear entirely as temperatures rise beyond their tolerance, the Ranch's Joshua tree woodlands appear to be expanding (Appelbaum et al., 2010; Cole et al., 2011). The combination of higher-altitude areas immediately adjacent to current Joshua tree range, and *Yucca brevifolia herbertii*'s unusual ability to survive and resprout after fire, may allow the Ranch to act as a refuge for this community in the coming decades (Appelbaum et al., 2010).

The fire regime in which this community evolved was a product of several interrelated factors, including elevation, topography, climate (both rainfall and lightning frequency), and the continuity and type of fuel cover (Brooks & Matchett, 2006). Due to the frequency of summer thunderstorms over mid-to-high elevation areas of the Mojave Desert—this area receives more lightning strikes per 100 km^2 than any other California bioregion—ignitions are relatively frequent (Brooks & Minnich, 2006). But in an environment dominated by large, widely-spaced plants, fire spread is severely limited by the abundance and continuity of ground-level fuel cover (DeFalco et al., 2009). Fuel cover is largely a product of the previous winter's rainfall, with rainy winters increasing the relative abundance and continuity of fine fuels, and therefore the size and frequency of summer fires. Elevation and topography affect rainfall, as air moving over mountains undergoes adiabatic cooling, but land features can also include physical barriers, such as abrupt ridges, that block or redirect the spread of fire (Brooks & Matchett, 2006). These factors also affect the seasonality of fire: with most precipitation occurring between November and April, and most thunderstorms occurring between July and September, fires are more likely in the summer and early fall than other times of year (Brooks & Minnich, 2006).

In much of the Mojave, these factors led to a regime of very infrequent, low severity fires (Brooks & Matchett, 2006). The pre-European FRI for desert mixed scrub has been estimated to range from 610 to 1,440 years (Safford et al., 2011). Recent FRIs for communities containing Joshua trees have been estimated to range from 35 to 100 years, although CAL FIRE records suggest that FRIs in the Ranch's deserts are longer.[44]

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[44] Gucker (2006) gives a number of post-settlement FRIs for plant communities containing Joshua trees, none of which is a perfect fit for the Ranch's Joshua tree woodlands. Those that depart from the 35 to 100 year range include basin big sagebrush (12 - 43 years), western juniper (20 - 70 years) and Colorado pinyon (10 - 400+ years) (Gucker, 2006). Cheatgrass-infested communities also depart from the 35 to 100 year range, with a fire return interval of <10 years (Gucker, 2006).

b. Fire Effects and Adaptations

As these FRIs suggest, fire has only recently become a controlling factor in natural selection, and the dominant vegetation in much of the Mojave (including *Yucca brevifolia*) is poorly adapted to it (Hereford et al., 2006). In Joshua Tree National Park, for example, *Yucca brevifolia* has been observed to experience high mortality during wildfires and low recruitment afterwards. Associated species are affected as well: biodiversity among rodents in burned areas of the Park is significantly lower than in unburned areas, and can remain so for years (Vamstad & Rotenberry, 2009).

In parts of the Western Mojave, fires appear to have been frequent enough to select for Joshua trees with the ability to resprout, and reproduce clonally from rhizomes, after being burned[45] (Barbour et al., 2007). It is far from clear, however, that *Yucca brevifolia herbertii* actually benefits from fire, and we have not found any evidence that other Mojave Desert species do. Moreover, given the underlying fuel dynamics, fires are unlikely to have been frequent or severe.

c. Changing Fire Regimes

Fire regimes in Joshua tree woodlands are changing due to four anthropogenic factors: 1) the increasing dominance of invasive annual grasses, which both cause and benefit from increases in the frequency and severity of wildfires; 2) the growth of fine fuels caused by anthropogenic nitrogen deposition; 3) human-caused ignitions; and 4) climate change (Brooks, 2003; DeFalco et al., 2009; Vamstad & Rotenberry, 2009). The most important of these by far is the spread of invasive grasses (Brooks, 2000; Brooks et al., 2011; Brooks & Matchett, 2006). Species such as cheatgrass (*Bromus tectorum*) and red brome (*Bromus madritensis* ssp. *rubens*), which expand in wet years and hold their ground in dry years, can create a continuous layer of fine fuel, which facilitates fire spread and increases severity (Brooks et al., 2011).

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[45] There is some controversy as to whether *Yucca brevifolia herbertii* should be considered a separate subspecies of Joshua tree or should be treated as synonymous with *Yucca brevifolia brevifolia*. Rowlands (1978) notes that the characteristics generally associated with *herbertii*—shorter stature and the ability to clone—are not perfectly correlated with this variety, and are sometimes seen in *Yucca brevifolia brevifolia*. In addition, cloning has been observed in some populations of *Yucca brevifolia jaegeriana*, a variety of Joshua tree occurring in the northeastern Mojave desert (Rowlands, 1978). But, as Barbour points out, "[t]he low, rhizomatous forms of *Y. brevifolia* corresponding to forma *herbertii* are widely known to resprout well following fire[,] . . . whereas other forms tend to either resprout weakly or are killed by fire." For this reason, while recognizing the "taxonomic fluidity of the species," this report will treat *Yucca brevifolia herbertii* as a separate variety of Joshua tree (Barbour et al., 2007).

Because invasive grasses are adapted to a high-frequency, high severity regime, and are often annuals instead of perennials, they regenerate more quickly than most native vegetation. This leads to a self-perpetuating cycle of more severe fire, followed by more abundant invasives (Brooks & Matchett, 2006; DeFalco et al., 2009). In parts of the Mojave, this cycle has advanced so far that invasive annual grasses form a majority of plant biomass (Brooks et al., 2011).

Nitrogen deposition and anthropogenic ignitions are likely to accelerate this process. Nitrogen deposition, some of which comes from airborne pollutants, makes previously nitrogen-poor desert soils more conducive to the spread of dense vegetation, a competitive advantage for fast-growing grasses (Brooks, 2003). Anthropogenic ignitions create additional opportunities for fire, thereby adding momentum to the cycle.

The effects of climate change are less clear, mainly because of uncertainty about future precipitation levels. If precipitation and temperature both increase, climate change is likely to add momentum to the invasive/wildfire cycle. Increased precipitation would produce more abundant and continuous fuel cover and, provided that there is still a dry season, increased temperatures would make this cover more flammable. But decreases in precipitation or changes in seasonality could lead to less abundant or less flammable fuel cover.

The invasive/wildfire cycle has not proceeded as far at Tejon as it has in other parts of the Mojave. But invasives such as *Bromus tectorum* are visibly present on this part of the Ranch (personal observation May 2011). As they gain ground, they are likely to facilitate the spread of fire, and ultimately make fire suppression more difficult (DeFalco et al., 2009). If they outcompete native species when suppression fails, then the invasive/wildfire cycle will gradually advance.

d. Management Approaches

1) Grazing

In the Mojave Desert, grazing can result in the trampling or preferential consumption of slow-growing desert vegetation, including species that shelter Joshua tree seedlings (Appelbaum et al., 2010; Brittingham & Walker, 2000; U.S. Fish and Wildlife Service, 2011b). It has also been observed to cause soil compaction and erosion. Because compacted desert soil can take over a century to return to its previous condition, and erosion of even a few centimeters can lead to "[g]ross disorganization of community structure," these are significant drawbacks (Lovich & Bainbridge, 1999; Webb, 2002; Webb & Stielstra, 1979).

Grazing has also been implicated in the destruction of desert tortoise (*Gopherus agassizii*) burrows (Lovich & Bainbridge, 1999; U.S. Fish and Wildlife

Service, 2011b). While Lovich and Bainbridge (1999) questioned the magnitude of this effect, the U.S. Fish and Wildlife Service has observed serious enough harm to justify restricting grazing on publicly-owned desert tortoise habitat[46] (U.S. Fish and Wildlife Service, 2011b). In addition to its effects on desert tortoises, a protective fencing experiment found that grazing, in conjunction with other disturbances such as off-highway vehicle use, was correlated with a loss of abundance and species richness among Mojave Desert birds and lizards (Brooks, 1999). For these reasons, grazing is unlikely to be a useful fire management tool in the Ranch's Joshua tree woodlands or other Mojave communities.

2) Mechanical Thinning

Manual and mechanical removal of invasive species is not advisable in sensitive desert ecosystems. Disturbance associated with mechanical treatments can make desert conditions even more severe, reducing infiltration and the moisture holding capacity of the soil (Lovich & Bainbridge, 1999).

3) Herbicides

Following a widespread fire in 2010, Joshua Tree National Park initiated a recovery program that included the use of herbicides for controlling brome grasses (Joshua Tree National Park, 2010). The long-term effects of herbicide use in Joshua tree woodlands remain unclear.

4) Revegetation

Revegetation has been tried in Joshua tree woodlands following destructive fires (Joshua Tree National Park, 2010). But the successful establishment of planted species is unreliable due to severe desert conditions, including high herbivory, high temperatures, intense sunlight, unpredictable rainfall, and low soil fertility (Lovich & Bainbridge, 1999). Moreover, Lovich and Bainbridge (1999) found direct seeding in desert ecosystems to be unsuccessful relative to transplants. If revegetation is used, target species should be planted in the cooler months of fall and spring and monitored on a regular basis to ensure successful establishment (Joshua Tree National Park, 2010).

10. Mojavean Scrub

The Ranch hosts over 16,700 acres of Mojavean scrub beyond its Joshua tree woodlands. Rather than having a single dominant species, these communities are

[46] Where possible, grazing will be restricted to the months (October 15 through March 15) when tortoises are inactive, or phased out entirely (U.S. Fish and Wildlife Service, 2011b).

characterized by a variety of widely-spaced cacti and shrubs, including beavertail (*Opuntia basilaris*), creosote bush (*Larrea tridentata*) and desert almond (*Prunus fasciculata*) (David Magney Environmental Consulting, 2010). Because of low surface fuel loads and low horizontal continuity, they rarely burn: according to one estimate, the presettlement median FRI for Mojave desert scrub was 610 years[47] (Brooks & Minnich, 2006; DeFalco et al., 2009; Safford et al., 2011). For this reason, desert scrub communities are among the least fire-adapted on the Ranch. As in Joshua tree woodlands, the main threat of type conversion comes from the invasive/wildfire cycle.

11. Saltbush Scrub

Saltbush scrub was once more prevalent in the San Joaquin Valley than it is today (Germano et al., 2001, Wills, 2006). The areas that remain on the Ranch are dominated by *Atriplex* species, including *Atriplex lentiformis* (big saltbush), *Atriplex polycarpa* (common saltbush), and *Atriplex spinifera* (spiny saltbush) (Tejon Ranch Company and Tejon Ranch Conservancy, 2009). Prior to the 1980s, fire management was of little concern in saltbush scrub ecosystems due to the lack of continuous fuels (Paysen et al., 2000, West, 1994). Montgomery and Cheo (1969) found that saltbush species are relatively slow burning because they have high moisture content and a layer of vesicular hairs that store water and prevent excessive transpiration. As such, saltbush species are believed to present less fire hazard than more flammable chaparral species. They have even been planted in fuel breaks in chaparral to stabilize soils and provide cover for wildlife (Nord et al., 1971).

Under natural conditions, fire is believed to occur every 35 to 100 years in saltbush scrub communities (Meyer, 2005). With the profusion of invasive annual grasses in desert shrublands, fire risk in saltbush scrub ecosystems has increased (Paysen et al., 2000). Yet the response of saltbush scrub species to fire has not been well documented. Saltbush species appear to have varied levels of fire tolerance and ability to recover. In a study conducted by Germano et al. (2001) in the San Joaquin Valley, it was found that only 0.2% of native saltbush species survived a fire event and that none of the dead species resprouted. Other species have been able to recover more quickly following a fire. *Atriplex lentiformis* and *Atriplex polycarpa*, which are present on the Ranch, produce large seedbanks and most likely establish following a fire by seed production rather than sprouting (Howard, 2003). However, no studies actually document the ability of seedbanks from saltbush species to colonize burn sites (Meyer, 2005). While more research is needed to adequately understand the nature of saltbush scrub recovery following fire—and this research can start in areas burned by the Comanche Fire of 2011—the profusion of flammable grasses in these systems does appear to be increasing fire risk (Meyer, 2005).

[47] Elsewhere in Southern California, a study of San Diego County fire records by vegetation type found that deserts had a burn cycle of 941 years (Wells et al., 2004).

APPENDIX B: ADDITIONAL BACKGROUND ON RANCHWIDE DRIVERS OF FIRE

1. Weather and Climate

Weather variables that affect fire behavior are collectively known as "fire weather," and include atmospheric moisture, air temperature, atmospheric stability, wind speed and direction, and clouds and precipitation (Van Wagtendonk, 2006):

- **Atmospheric moisture** affects fuel moisture. Water vapor is exchanged between the atmosphere and fuels until equilibrium is reached. High atmospheric moisture results in higher fuel moisture, which reduces the flammability of the fuel.
- **Air temperature** affects fuel temperature. As air temperature increases, fuel temperature increases, which reduces the amount of additional energy needed to raise the fuel temperature to the ignition point.
- Under **unstable atmospheric conditions**, fires create convective columns that cause indrafts and strong surface winds.
- **Wind speed and direction** affect fire behavior in several ways. Wind dries out fuels and increases the supply of oxygen, contributing to more rapid combustion. Wind also facilitates fire spread by transferring heat, bringing flames in contact with fuel, and carrying embers.
- **Clouds** reduce air temperature, which lowers fuel temperature. **Precipitation** raises fuel moisture. Both effects inhibit fire spread.

Fire behavior is driven by fire weather in the short term and climate in the long term. Average precipitation, average temperature, and other climate variables have a profound influence on plant growth and hydrologic cycling, which in turn affect fuel loads and fuel moisture (Minnich, 2006).

Precipitation influences fire through its duel effects on fuel availability and flammability. While increased precipitation decreases fuel moisture and flammability, it also stimulates plant growth, which leads to greater fuel availability. The net effect depends on other climatic and ecological factors. In moist areas with dense vegetation, changes in flammability tend to dominate. In arid areas with sparse vegetation, changes in fuel availability have the greatest impact on wildfire occurrence (Westerling, 2008).

a. Weather and Climate at Tejon Ranch

With elevations ranging from 142 to 2017 meters, Tejon Ranch has a complex and varied climate. At low elevations, average annual maximum temperature reaches a peak of approximately 35° C in July and August, while at higher elevations the average annual maximum can be up to 5° C lower. Annual average minimum temperatures can dip below -1° C in the winter months (Western Regional Climate Center n.d.; The University of Utah, 2012). Over the past 50-100 years, higher elevation areas of Tejon Ranch have experienced a warming trend, while lower elevation areas have experienced a cooling trend (Appelbaum et al., 2010).

The region has a relatively arid climate: annual precipitation averages approximately 11-12 inches per year and occurs primarily during the winter months (Western Regional Climate Center, n.d.; Appelbaum et al., 2010). Organic matter decomposes slowly in such a climate, which results in relatively high fuel loads. Like other areas of southern California, the Ranch experiences a distinct fire season beginning in early summer (Minnich, 2006).

b. Climate Change

Future climate change—manifested as changes in patterns of temperature, precipitation, wind, flooding, drought, and other variables—may alter fire regimes through effects on plant growth, fuel moisture, rate of fire spread, ignition frequency, or other factors. Climate change is expected to result in higher mean annual temperatures in southern California. The PCM-A2 scenario, for example, predicts a 2.5° C increase in mean annual temperature in the region, while the GFDL-A2 scenario predicts a 4.4° C increase (Cayan et al., 2008).

The effects of climate change on fire regimes in southern California will depend heavily on the direction and magnitude of precipitation changes (Westerling, 2008). However, there is significant uncertainty as to whether mean annual precipitation will increase or decrease. For example, the PCM-A2 scenario predicts an 8% increase in mean annual precipitation, while the GFDL-A2 scenario predicts a 26% decrease (Cayan et al., 2008). Due to the region's semi-arid climate, changes in precipitation will likely affect fire behavior primarily through changes in fuel availability (rather than changes in flammability).

Climate change is expected to cause a 9-15% increase in the total acreage burned annually in California. The amount of biomass burned is also expected to increase, at least initially. Under the PCM-A2 scenario, the quantity of biomass burned is predicted to be 18% greater by 2100 (due to increased net primary productivity (NPP)). Under the GFDL-A2, biomass burned is predicted to increase initially (due to decreased fuel moisture), but decrease by the end of the century (due to reduced NPP) (Lenihan et al., 2008).

2. Development

Tejon Ranch is 60 miles north of Los Angeles and 30 miles south of Bakersfield. Areas around the Ranch have relatively low population density, but are growing. Kern County has approximately 840,000 residents, a 27% increase over 2000 levels (US Census, 2010). The largest neighboring city is Arvin, which has a population of 19,300 (US Census, 2010).

Several studies have shown a strong positive correlation between population density and fire. Cardille et al. (2001) and Keeley et al. (1999) found that areas with higher population densities are more prone to fire ignitions. Yet Syphard et al. (2007) caution against general conclusions about the relationship between population density and fire frequency. In their 2006 study, Syphard et al. found that intermediate levels of urbanization, when compared to high and low levels of development, are actually correlated with the highest number of ignitions due to the spatial arrangement of developments. Aggregated patterns of predicted growth resulted in only a minimal increase in WUI and no increase in ignitions (Syphard et al., 2006). Syphard defines intermediate housing density as 49 to 742 housing units per km^2 and intermediate levels of population density as 35 to 45 people per km^2 (Syphard et al., 2007).

At Tejon Ranch, the Centennial development is expected to reach a maximum of 485 housing units per km^2. The twenty-year development plan indicates that 1,000 homes will be built annually (Centennial California, n.d.). The projected housing density in Centennial falls within the range which Syphard et al. (2007) define as intermediate and is correlated with the highest number of fire ignitions. Tejon Mountain Village anticipates 32 housing units per km^2 which is on the lower range of development levels[48] (Dudek, 2009b).

While human development patterns have been shown to influence the location and number of fire ignitions in California, fire spread and frequency is mainly a function of vegetation characteristics (Syphard et al., 2007; Syphard et al., 2008). In southern California, more fires actually occurred at greater distances from development where there is more continuous vegetation (Syphard et al., 2008). In other words, fire ignitions may occur closer to roads and developments but the areas that burn the most frequently are non-urban areas where fire spreads after ignition (Syphard et al., 2008). Fire spreads best across a landscape that is approximately

[48] The Tejon Mountain Village Fire Protection Plan (Dudek, 2009b) has outlined in great detail a number of strategies for managing wildfire risk. For example, buildings in the new community will be constructed out of fire resistant materials and will have added on-site water availability for firefighters. Firebreaks or "asset protection zones" will be positioned around housing complexes, allowing a cleared strip of land to act as a defensive perimeter and access point for firefighters. These zones can be effective in minimizing ignitions caused by radiation from flames, though wind-blown embers may still cross firebreaks (Gill & Stephens, 2009). Moreover, structures within TMV are required to have 100-foot-wide fuel modification zones (FMZs) where all brush will be removed, and areas adjacent to more volatile fuel types will have up to 200-foot- wide FMZs.

60% vegetative cover. Below 30% vegetative cover, loss of connective fuels can limit fire spread (Syphard et al., 2006). In order to manage for fire risk, managers need not only consider areas at high risk of ignition, but also areas where fires are most likely to spread.

3. Industry, Infrastructure and Resource Extraction

The Ranch hosts several industrial operations, and an extensive network of energy and telecommunications infrastructure. These include a limestone plant operated by long-term lessee National Cement, a 750-megawatt natural gas plant, and a number of pipelines, transmission lines and fiber optic cables (Tejon Ranch Company, n.d.). The Ranch also grants leases for oil and gas exploration. Given the need to protect Company and lessee assets, the areas of the Ranch devoted to these uses will almost certainly be subject to vigorous fire suppression as long as they remain in use.

But these activities may still increase fire risk in surrounding areas. Storage and transport of flammable hydrocarbons, for example, creates the risk of ignitions due to spills, particularly in areas where above-ground infrastructure is vulnerable to lighting or below-ground infrastructure is vulnerable to earthquakes (Renni et al., 2010). Similarly, strong winds can cause power lines to collide with trees or each other, throwing sparks into surrounding vegetation. When wind speeds are high and fuel moisture is low, sparks from a transmission line can lead to a major conflagration (Tse & Fernandez-Pello, 1998).

4. Recreation

As recreational use of outdoor areas increases, anthropogenic ignitions are likely to increase as well (Syphard, 2007). Activities that can drive ignitions often occur in remote areas that are more difficult for emergency crews to access, which allows fires to build and spread out of control (Stephens, 2005). These activities include hunting, camping, and the use of motor vehicles on backwoods trails.

Despite a temporary moratorium, hunting is likely to remain an important activity on the Ranch, and has the potential to act as a source of ignitions. Firearms emit open flames, hot gases, embers, and metal shells that can spark fires in dry conditions (Babrauskas, 2005). Additionally, the friction caused by a bullet striking a hard surface can occasionally spark a fire, as happened in the Angeles National Forest (Haston et al., 2009).

Recreational travel can also contribute to ignitions. Vehicle travel in areas overgrown with tall brush can start fires in at least two ways: 1) emission of hot particles from tailpipes; and 2) direct contact of vegetation with a vehicle's exhaust system (Babrauskas, 2005). The increasing incidence of these activities on the Ranch may therefore cause more frequent ignitions.

5. Invasive Plant Species

a. Postfire Colonization and Effects of Invasives on Fire Regimes

Postfire invasive colonizers tend to be prolific reproducers through seed production, vegetative reproduction, or both. Depending on fire severity, part of the plant usually survives, either by being underground or by being protected by thick bark. The typical invasive plant is also a self-pollinator, has low shade tolerance, and a short generation time (usually biennial or annual). These conditions make them prime candidates for postfire colonization. Invasibility is largely influenced by propagule pressure, which can come from an existing seedbank, or from the plants surrounding a burn area (Zouhar et al., 2008).

Once established, invasive populations affect fuel structure significantly. Each plant has both intrinsic fuel properties, such as chemical volatility and moisture, and extrinsic properties, which relate to the landscape. The chemical composition of plants can affect fire either by providing volatile compounds or by contributing to a more rapid decomposition, although invasive plants have not been observed to differ from natives in this regard. The *Bromus* and *Avena* species are examples of invasives with strong extrinsic fuel properties. Grasses, in particular, increase stocks of fast-drying, standing dead fuels. Finely textured grasses can extend the fire season because they dry out before natives, thereby starting the fire season earlier (Brooks et al., 2011).

b. Effects of Fire on Invasives

Fire can promote invasive species. Most obviously, fire is a major ecosystem disturbance, and opens up burned areas for colonization by invasive plants. There are also subtler fire effects that make burned areas easier for invasives to colonize. For example, fire is an important part of long term nutrient cycling. The postfire period is often characterized by an increase in nutrients such as nitrogen and phosphorus, both of which are normally limiting nutrients for plants. Generalist weedy annuals take advantage of increases in these nutrients more quickly than native perennials, especially in low nutrient soils such as deserts (Brooks & Pyne, 2001). In addition, the rate of invasion can be affected by fire intensity (Hunter et al., 2006).

c. Invasive Plants on Tejon Ranch

The 32 most damaging invasive plants have been identified and surveyed across Tejon Ranch (Knapp, 2010). Some, such as tamarisk and *Arundo donax*, are known to have significant impacts on fire regimes. Others benefit from fire, but do not themselves have a dramatic effect on fire regimes. Table 10 summarizes the top five invasive plants' relationship with fire.

Plant Species	**Net Area (ft^2)**	**Effects on fire**	**Fire effects on plant**
Hirschfeldia incana (summer mustard)	2,401,086	Increases fuel loads where nonnative annual grasses have altered the fire regime. May increase fire intensity but effect is minor (Cal-IPC, 2006).	A high fire frequency (or any disturbance) leads to a spread in the plant.
Salsola tragus (Russian thistle)	238,602	Increases fire hazard, especially along fences. Long taproots extract deep soil moisture. Stem spacing for air circulations and slow decomposition makes it burn easily. It also carries fire by rolling across the landscape.	Colonizes quickly after fire. The seed is spread quickly. It will colonize within 1-3 years (Howard, 1992b).
Silybum marianum (milk thistle)	609,898	N/A	Fire increases spread of milk thistle, as does overgrazing. Toxic to cattle and sheep (Bean, 2006).
Marrubium vulgare (horehound)	509,043	N/A[49]	Fire intolerant (Esser, 1993).
Eucalyptus sp.[50]	250,205	Produces large amounts of debris which spread spot fires. Increases fire intensity and frequency. Plant contains volatile oils which create a hot fire.	Leaves and seeds are fire tolerant. After a fire, the trees shed seeds at a faster rate (Esser, 1993).

Table 10: Relationship between five invasives and fire regimes.

[49] Horehound might increase fire intensity by increasing biomass. However, this was only mentioned in an Australian evaluation of the plant, and invasive plants can have very different effects depending on location.

[50] The species of Eucalyptus was not identified. The effects listed are for *Eucalyptus globules*, which is known to have more severe fire effects than other species.

APPENDIX C: BACKGROUND ON FRID ANALYSIS AND COMMUNITY-SPECIFIC FRID MAPS

1. Historical Fire Return Intervals Used in FRID Analysis

Ranch Vegetation Community	CALVEG Classifica-tions	Fire Regime Group	Median FRI	High FRI	# Depart. for which Median Is still in Historic Range	Source
Alkali Meadow	N/A	Grassland	3	8	2	Stephens et al., 2007
Alluvial Scrub	CALVEG Not Used	Desert Mixed Scrub	610	1440	2	Safford et al., 2011
Annual Grassland	N/A	Grassland	3	8	2	Stephens et al., 2007
Black Oak Savannah	Black Oak	Yellow Pine	7	40	5	Safford et al., 2011
Black Oak Woodland	Black Oak	Yellow Pine	7	40	5	Safford et al., 2011
Blue Oak Savannah	Blue Oak	Oak Woodland	12	45	3	Safford et al., 2011
Blue Oak Woodland	Blue Oak	Oak Woodland	12	45	3	Safford et al., 2011
Brewers Oak Scrub	CALVEG Not Used	Chaparral and Serotinous Conifers	59	90	1	Safford et al., 2011
Burrobrush Scrub	CALVEG Not Used	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
California Buckeye Woodland	California Buckeye	Mixed Evergreen	13	80	6	Safford et al., 2011
Canyon Oak Savannah	Canyon Live Oak	Mixed Evergreen	13	80	6	Safford et al., 2011
Canyon Oak Woodland	Canyon Live Oak	Mixed Evergreen	13	80	6	Safford et al., 2011
Chaparral	Southern Mixed Chaparral, Manzanita Chaparral, Ceanothus Mixed Chaparral	Chaparral and Serotinous Conifers	59	90	1	Safford et al., 2011
Conifer/Mixed Oak	Ponderosa Pine - White Fir	Dry Mixed Conifer	9	50	5	Safford et al., 2011
Digger Pine Savannah	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Digger Pine Woodland	Gray Pine	Oak Woodland	12	45	3	Safford et al., 2011

Disturbed / Nonnative Grassland	N/A	Grassland	3	8	2	Stephens et al., 2007
Grassland	N/A	Grassland	3	8	2	Stephens et al., 2007
Incense Cedar Stand	Incense Cedar	Dry Mixed Conifer	9	50	5	Safford et al., 2011
Interior Oak Savannah	N/A	Oak Woodland	13	80	6	Safford et al., 2011
Interior Oak Woodland	Interior Live Oak	Mixed Evergreen	13	80	6	Safford et al., 2011
Intermixed Conifer	Ponderosa Pine - White Fir	Dry Mixed Conifer	9	50	5	Safford et al., 2011
Isomeris Scrub	CALVEG Not Used	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
Joshua Stand	Joshua Tree	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
Joshua-Juniper Woodland	Joshua Tree	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
Mixed Oak Savannah	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Mixed Oak Woodland	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Mojavean Scrub	CALVEG Not Used	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
Native Grassland	N/A	Grassland	3	8	2	Stephens et al., 2007
Oak Savannah	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Oak Woodland	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Pinyon Pine Woodland	Singleleaf Pinyon Pine	Pinyon Juniper	94	250	2	Safford et al., 2011
Saltbush Scrub	Saltbush	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
Saltbush / Buckwheat Scrub	Saltbush	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
Scrub	Desert Mixed Shrub	Desert Mixed Shrub	610	1440	2	Safford et al., 2011
California Scrub Oak	Scrub Oak	Chaparral and Serotinous Conifers	59	90	1	Safford et al., 2011
Undetermined Chaparral	Southern Mixed Chaparral, Manzanita Chaparral, Ceanothus Mixed Chaparral	Chaparral and Serotinous Conifers	59	90	1	Safford et al., 2011
Undetermined Savannah	N/A	Oak Woodland	12	45	3	
Undetermined Woodland	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Valley Saltbush	Saltbush	Desert Mixed	610	1440	2	Stephens

Scrub		Shrub				et al., 2007
White Fir Stand	Ponderosa Pine - White Fir	Dry Mixed Conifer	9	50	5	Safford et al., 2011
White Fir / Mixed Oak	Ponderosa Pine - White Fir	Dry Mixed Conifer	9	50	5	Safford et al., 2011
White Oak Savannah	N/A	Oak Woodland	12	45	3	Safford et al., 2011
White Oak Woodland	N/A	Oak Woodland	12	45	3	Safford et al., 2011
Wildflower / Annual Grassland	N/A	Grassland	3	8	2	Stephens et al., 2007

Table 11: Fire return intervals used in FRID analysis.

2. Sources of Modern FRIs Used in FRID Analysis

We used the following sources to build a spatially explicit picture of the Ranch's fire history: 1) CAL FIRE perimeter data, which goes back to 1878 for timber fires greater than 10 acres, brush fires greater than 50 acres, and grass fires greater than 300 acres; 2) perimeter data from the Ranch for the controlled burns conducted in 1987, 1988, 1989 and 1992; and 3) Geospatial Multi-Agency Coordination Group (GeoMAC) perimeter data for the Comanche and Keene Fires of 2011.

3. Community-Specific FRID Maps

The following maps highlight the results of our FRID analysis in particular ecological communities:

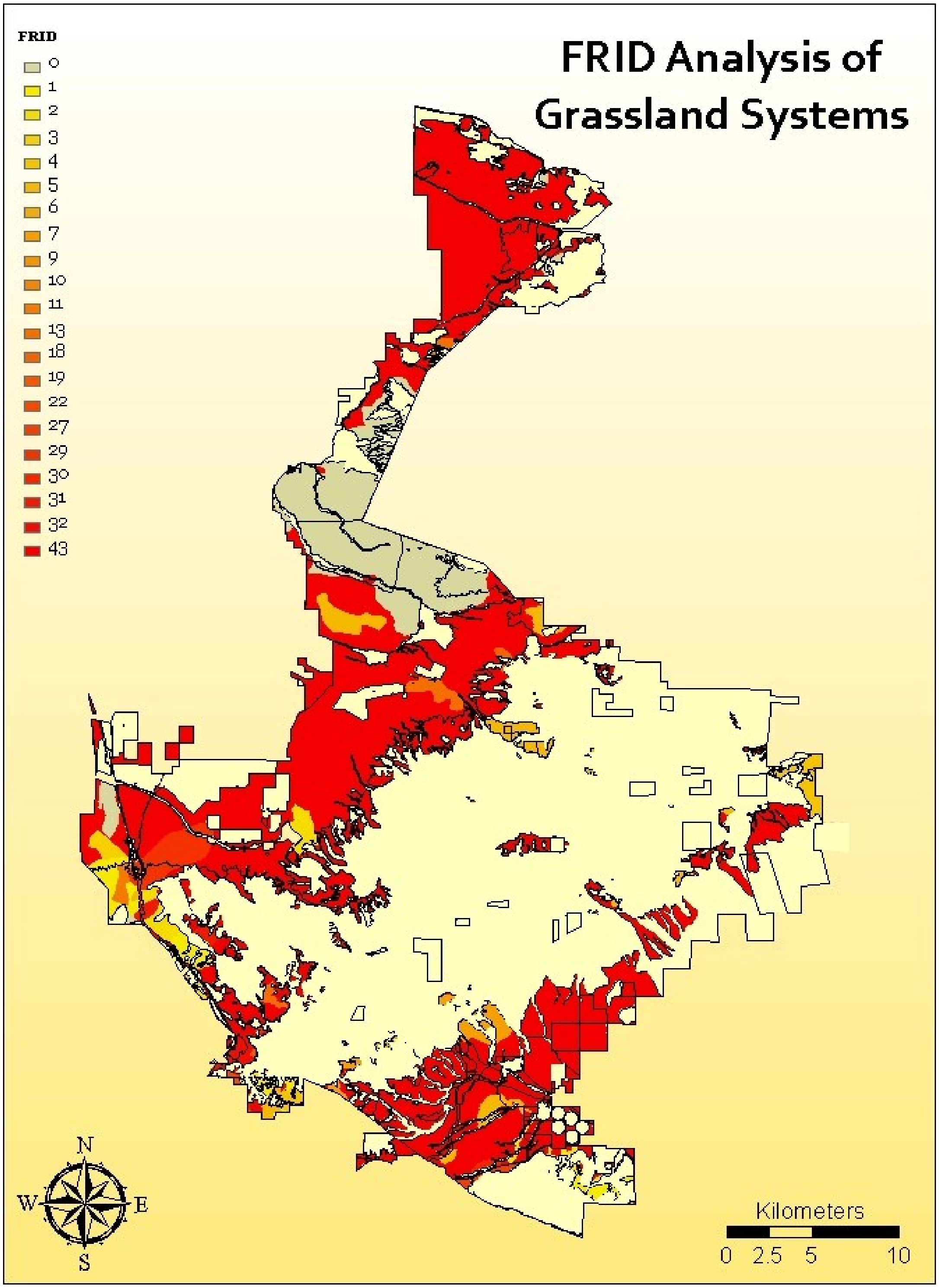

Figure 18: Median FRID map highlighting grassland systems.

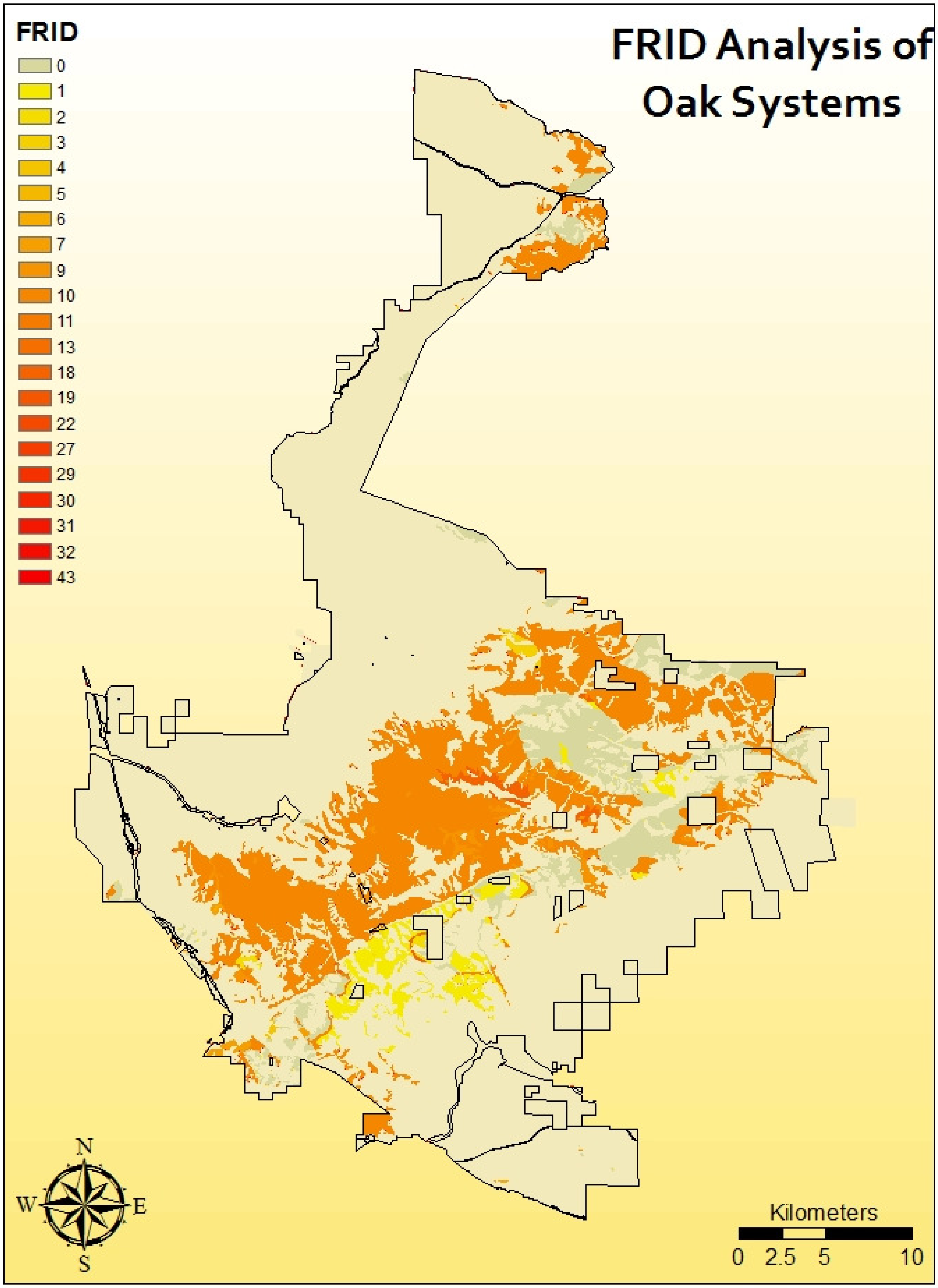

Figure 19: Median FRID map highlighting oak systems.

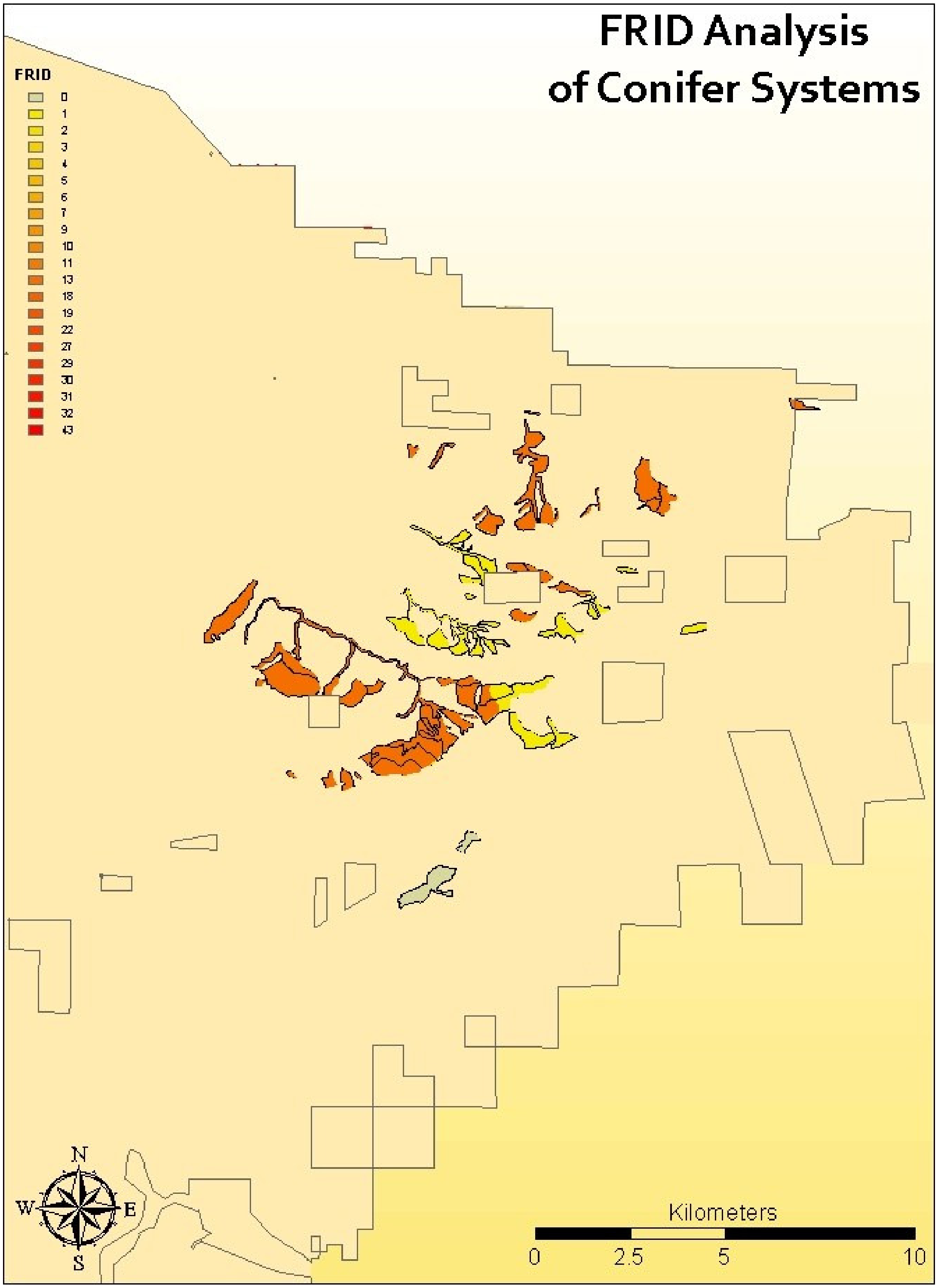

Figure 20: Median FRID map highlighting conifer systems.

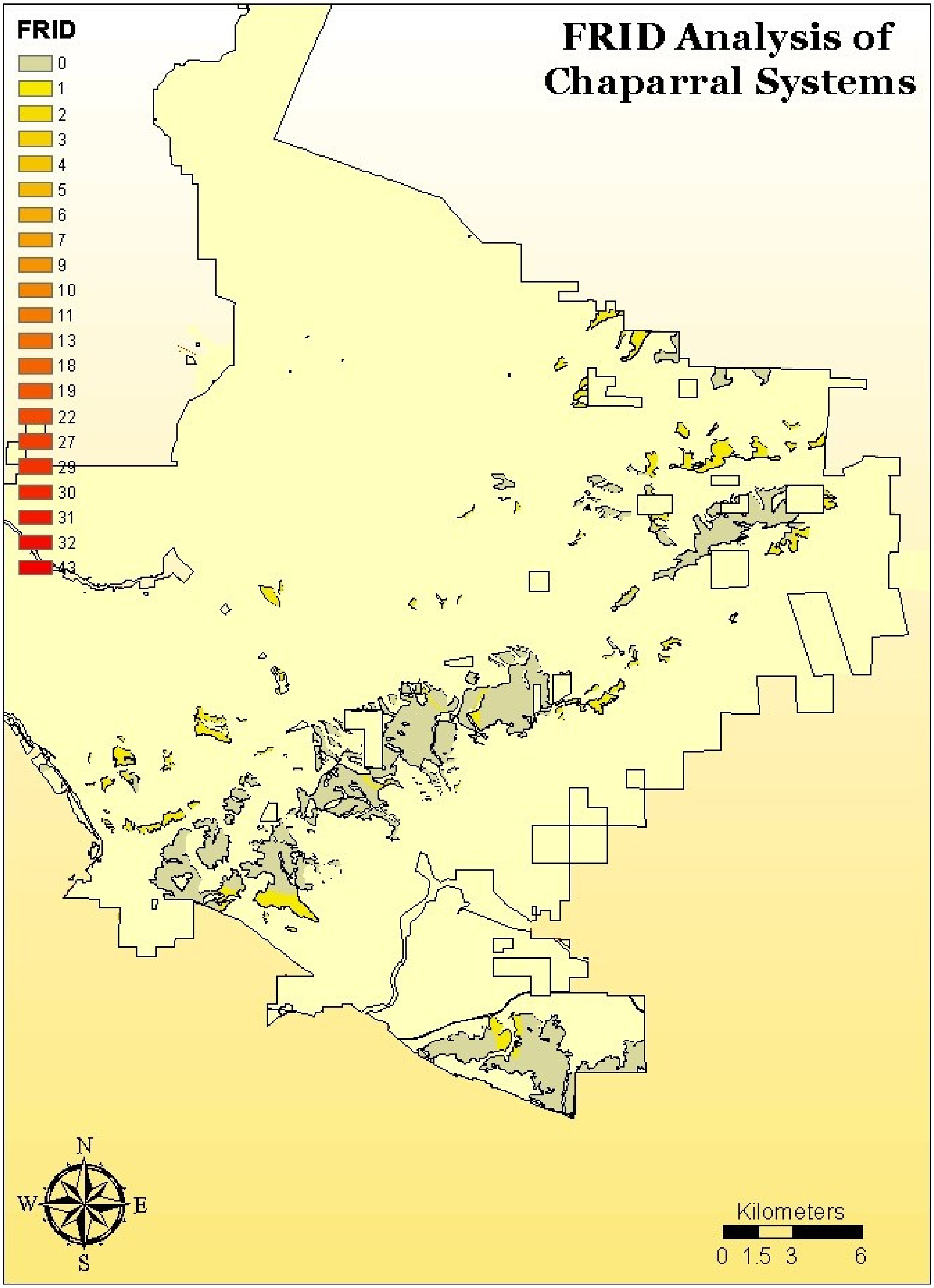

Figure 21: Median FRID map highlighting chaparral systems.

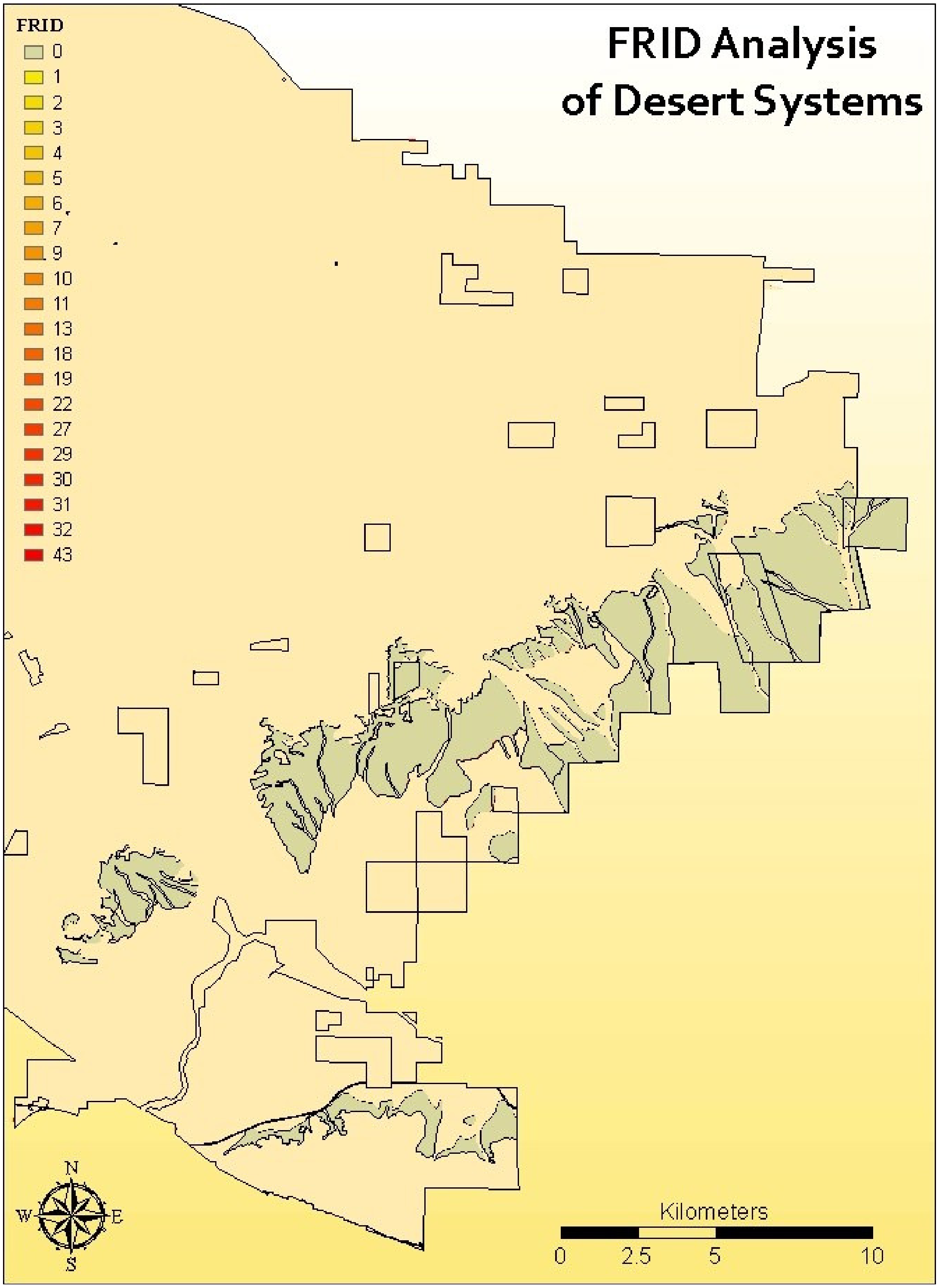

Figure 22: Median FRID map highlighting desert systems.

APPENDIX D: FULL DESCRIPTION OF LANDIS-II METHODS AND ADDITIONAL LANDIS-II RESULTS

1. Input Files

a. Species

Species parameter values were obtained from a variety of sources, such as the USFS Silvics Manual (Burns & Honkala, 1990) and the USFS Fire Effects Information System (see Table 12).[51] Decay rates were taken from Pastor and Post (1986) for general classes of large, medium, and small wood.

Scientific Name	Long-evity	Age Sex. Mat.	Shade Tol.	Fire Tol.	Effect. Seed Disperse Dist.	Max. Seed Disperse Dist.	Veg. Repro. Prob.	Sprout Min Age	Sprout Max Age	Post-Fire Regen.
Quercus berberidifolia (Scrub Oak)	120	40	5	4	60	100	0.9	0	120	resprout
Adenostoma fasciculatum (Chamise)	150	3	2	1	5	10	0.7	3	100	resprout
Arcto-staphylos glauca (Bigberry Manzanita)	100	20	3	2	60	100	0	0	100	serotiny
Ceanothus cordulatus (Mountain Whitethorn)	50	20	2	2	60	300	0	0	50	serotiny
Grass spp.	10	1	1	1	100	100	0	0	0	none
Pinus ponderosa (Ponderosa Pine)	300	7	2	5	37	120	0	0	0	none
Calocedrus spp. (Incense Cedar)	500	40	4	4	60	100	0	0	0	none
Abies concolor (White Fir)	300	40	4	4	60	100	0	0	0	none
Quercus kelloggii (Black Oak)	300	30	3	2	30	100	0.8	1	300	resprout

[51] The full list of references includes: Ackerly, 2004; Barbour et al., 2007; Burns & Honkala, 1990; Ewers & Schmid, 1981; Franklin et al., 2005; Hickman, 1996; Jow et al., 1980; Keeley & Keeley, 1981; Mahall et al., 2010; Mediavilla et al., 2001; Rundel 1980; Sidahmed et al., 1982; Spencer et al., 2008; Syphard and Franklin, 2004; U.S. Forest Service Fire Effects Information System; USDA Natural Resources Conservation Service, 2012; U.S. Forest Service, 2011.

Quercus chrysolepis (Canyon Live Oak)	250	20	3	1	30	100	0.95	1	250	resprout
Quercus wislizeni (Interior Live Oak)	200	20	3	1	30	100	0.8	1	200	resprout
Prunus emarginata (Bitter Cherry)	35	2	3	3	5	10	0	0	0	resprout
Quercus garryana (Brewers Oak)	100	20	2	4	30	100	0.6	1	100	resprout
Larrea tridentate (Creosote)	1000	10	5	2	1	1	0.9	0	0	resprout
Pinus sabiniana (Digger Pine)	200	18	1	3	30	100	0	0	0	none
Cercocarpus spp. (Mountain Mahogany)	150	5	2	2	50	500	0.95	3	150	resprout
Symphori-carpos spp. (Snowberry)	40	3	2	4	5	30	0.8	3	40	resprout
Quercus lobata (Valley Oak)	300	30	2	1	30	100	0.8	1	100	resprout
Quercus douglasii (Blue Oak)	250	20	3	1	30	100	0.8	1	250	resprout

Table 12: Species parameter values.

b. Ecoregions

Ecoregions in LANDIS-II are areas that represent relatively homogenous environments in terms of soils, climate, topography, and other abiotic and biotic characteristics. A total of 18 ecoregions were used, including 8 ecoregions inside the ROI and 10 inside the 2 km buffer. Ecoregions were designated based on a combination of elevation and USFS subsections. A total of 5 USFS subsections fell within the modeling area (ROI + buffer). One of these subsections occupied a very small portion of the modeling area, and was thus merged with an adjacent subsection. The ecoregions were further divided according to elevation using five equal-interval elevation ranges within the modeling area.

c. Initial Communities

The initial communities file defines the major vegetation types, identifies the dominant species found in each vegetation type, and specifies the age cohorts for each species present. Major vegetation types were based on a GIS vegetation map compiled from a variety of vegetation surveys conducted on the Ranch since 1980.

Dominant tree species for each vegetation community were determined primarily from the 1980 Timber Index Survey and the 1930s Wieslander Vegetation Type Mapping (VTM) plots. Additional information was drawn as necessary from CALVEG mapping and other sources.

d. Biomass Succession

The Biomass Succession file contains information needed to model succession at each timestep based on age cohort, biomass, and species attributes (Scheller & Mladenoff 2004). The extension produces maps and spreadsheets of biomass for each species at each timestep. Values for minimum relative biomass by shade class were taken from Figure A in Scheller & Mladenoff (2004).

Actual evapotranspiration (AET) was calculated from the MODIS Global Terrestrial Evapotranspiration Data Set (MOD16) annual products (Numerical Terradynamic Simulation Group, 2011). Average annual AET values were calculated for the entire ROI for 2000-2010 and then averaged across all years.

The establishment probabilities, maximum biomass, and maximum annual net primary productivity (Max ANPP) change at every timestep. Probabilities of species establishment by ecoregion were estimated from probabilities of species occurrence derived from a maximum entropy (MaxEnt) model. MaxEnt predicts species' geographic distribution based on both known species presence data (samples) and the environmental conditions at sites of known occurrence (features).[52] It estimates the maximum entropy probability distribution over all pixels in a region of interest (Philips et al., 2006). Species' presence locations were represented by a random point distribution within vegetation types where the species are known to occur. Geographic distributions were projected for three climate scenarios: current conditions, GFDL projections averaged over the years 2041 to 2070, and PCM projections averaged over the years 2040 to 2069. Species' establishment probabilities were estimated by averaging species' occurrence probabilities for pixels occurring within each ecoregion.

Maximum aboveground net primary productivity[53] and maximum biomass for each ecoregion were calculated using the Forest Vegetation Simulator (FVS) (U.S. Forest Service, 2011). Southern California Forest Inventory and Analysis plots were selected which represented the elevation and vegetation communities on Tejon. FVS

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[52] Environmental conditions were represented by an aridity index, growing degree days above 5° C, minimum temperature of the coldest period, mean annual precipitation, temperature seasonality, available water holding capacity of soil, and soil pH.

[53] The LANDIS-II user guide terms ANPP as Annual Net Primary Productivity but it is clear from the description the actual parameter is Annual Net Primary Production.

was then used to calculate ANPP and maximum biomass for each species in each ecoregion under current climate and projected future climates. Shrub and chaparral species data were not available in FVS and so were taken from the general literature.

e. Dynamic Fire

The Dynamic Fire extension models fire ignition and spread across the landscape (Sturtevant et al. 2009). Fire spread is determined by weather, topography, and fuel type. The extension produces maps of time of last fire, percent dead fir, percent conifer, and fire severity at each time step. Two spreadsheets are also produced which detail each fire that has been modeled on the Ranch.

The table of ecoregion-dependent values includes ecoregion-specific parameters that define the fire size distribution, number of ignitions, and high/low foliar moisture content (FMC) by season. Mu and sigma for the fire size distributions were derived from fire history statistics calculated for 1950-2011 (California Department of Forestry and Fire Protection, 2012). FMC values were taken from Spencer et al. (2008).

In the seasons table, the proportion of fires occurring in each season was also calculated from fire history statistics for 1950-2011. The start of seasons were calculated by averaging the USGS Normalized Difference Vegetation Index (NDVI) data across the ROI over 5 years to give average values for start of season, end of season, and maximum NDVI. In the case of the spring start date, the NDVI-derived date was averaged with the date derived from the Canadian Forest Fire Weather Index (FWI) System (the third consecutive day with noon temperatures above 12° C) (Lawson & Armitage, 2008). The percent of grass curing was estimated based on data from the National Fuel Moisture database.

Representative daily weather records are provided in a dynamic weather table (not shown). Representative weather records were calculated based on weather data (temperature, relative humidity, wind speed, wind direction, and precipitation) from 2009-2011 from two Remote Automatic Weather Stations (RAWS) near the ROI (University of Utah, 2012). The weather table includes values for fine fuel moisture content, buildup index, wind speed, and wind direction. Fine fuel moisture content and buildup index were calculated using the FWI System (Loudermilk, 2011). Slope and aspect maps were also input into LANDIS in order for the fire modeling to incorporate topography.

f. Dynamic Fuels

The Dynamic Fuel extension uses base fuel types defined in the Canadian Forest Fire Behavior Prediction (FBP) System (Syphard et al. 2011). The fuel parameters were taken from Spencer et al. (2008). For each cell on the landscape, a score is computed for each fuel type depending on the species and age cohorts present, and then the fuel type with the highest score is assigned to each cell. The extension then produces a map of the fuel types at each time step.

g. Biomass Harvest

The Biomass Harvest extension was used to simulate management prescriptions in conifer systems. A single management area was designated, encompassing approximately 314 hectares along ridgetops and south-facing slopes dominated by mixed conifer, white fir, and incense cedar stands. The management area was divided into seven stands, which were randomly selected for treatment. Two management prescriptions were evaluated: 1) hand-thinning and 2) hand-thinning plus prescribed burning ("combined prescription"). Stands were classified as eligible for treatment if they were at least 50 years old, had not been treated within the past 20 years, and had at least 50 percent of cells occupied by ponderosa pine, white fir, or incense cedar.

Both management prescriptions were simulated by removing biomass from treated stands. In both cases, percentage biomass removed was inversely related to age cohort. For hand thinning, only biomass from tree species was removed. For the combined prescription, biomass from all species (including grass and shrubs) was removed. Figure 23 shows the percent biomass reduction of tree species by age cohort in the management areas for both management prescriptions. All treated areas were then classified into post-treatment fuel types for the next fifteen years.

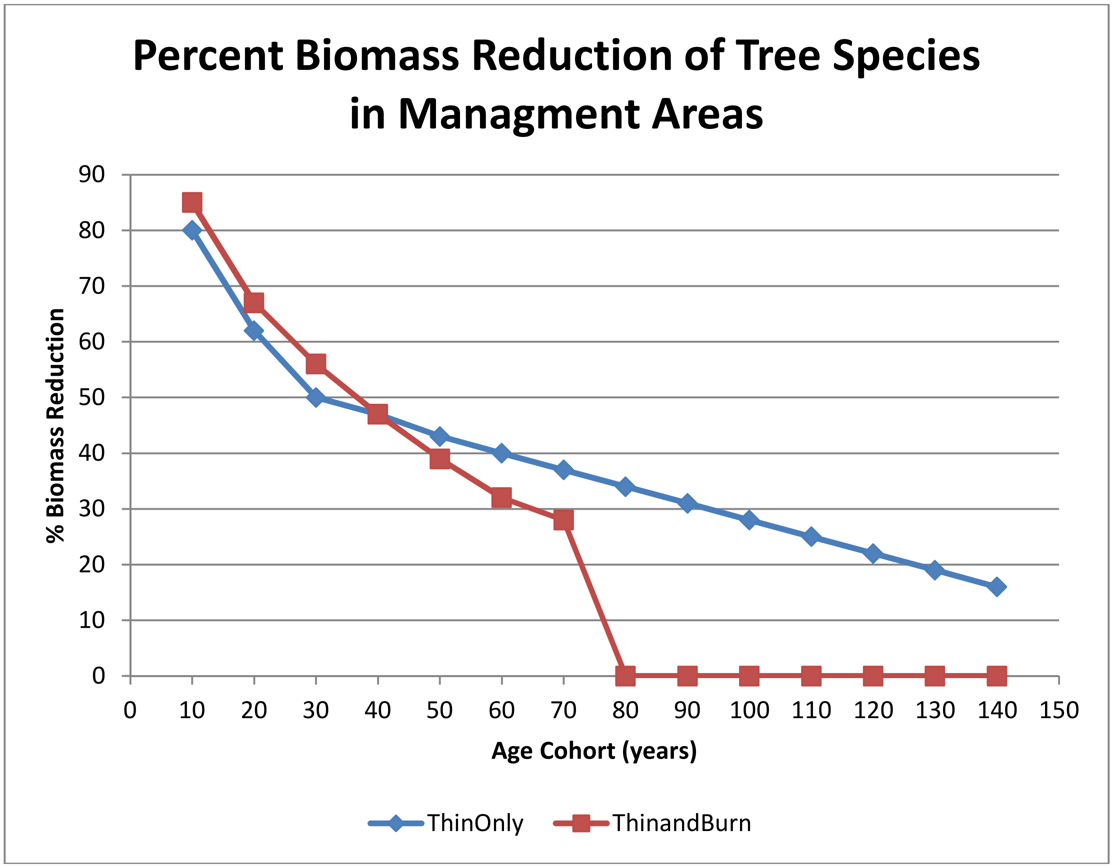

Figure 23: Percent biomass reduction of tree species in management areas.

2. Model Calibration

The model was calibrated so that vegetation communities were classified into the desired fuel type. Establishment probabilities derived directly from MaxEnt led to greater-than-expected spread of vegetation on Tejon. This is most likely because the MaxEnt output, probability of occurrence, is only a proxy for establishment probability. Additionally, LANDIS-II does not factor climate or topography into vegetation spread, which in reality would lead to different rates of spread in different directions. To calibrate the model, establishment probabilities were reduced to reflect the expected spread of species.

The fire and fuel extensions were calibrated by systematically adjusting the maximum fire size and the ignition probability until the output fell within ten percent of the expected FRI and average fire size for each ecoregion. The fuel extension classifies each cell into a particular fuel type based on the species that are listed for each fuel type. In each fuel type, the list of species was adjusted until the model classified the vegetation into the appropriate fuel type.

3. Ordered Difference results for ANOVA's comparing all scenarios

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
PCM,TB	C,TB	0.0624375	0.0234296	-0.010293	0.1351684	0.1610	
PCM,TB	C,N	0.0548135	0.0230198	-0.016645	0.1262724	0.2947	
PCM,TB	C,T	0.0528869	0.0228609	-0.018079	0.1238526	0.3342	
PCM,TB	PCM,T	0.0494871	0.0221816	-0.019370	0.1183440	0.3856	
PCM,TB	GDFL,TB	0.0477842	0.0226000	-0.022372	0.1179399	0.4635	
PCM,TB	GDFL,N	0.0466329	0.0227465	-0.023978	0.1172435	0.5082	
PCM,TB	GDFL,T	0.0424682	0.0225292	-0.027468	0.1124041	0.6242	
PCM,TB	PCM,N	0.0315942	0.0227279	-0.038958	0.1021468	0.9018	
PCM,N	C,TB	0.0308433	0.0234655	-0.041999	0.1036856	0.9273	
PCM,N	C,N	0.0232193	0.0230563	-0.048353	0.0947916	0.9853	
PCM,N	C,T	0.0212927	0.0228977	-0.049787	0.0923725	0.9913	
GDFL,T	C,TB	0.0199693	0.0232731	-0.052276	0.0922144	0.9949	
PCM,N	PCM,T	0.0178929	0.0222195	-0.051082	0.0868674	0.9967	
PCM,N	GDFL,TB	0.0161900	0.0226372	-0.054081	0.0864612	0.9986	
GDFL,N	C,TB	0.0158046	0.0234836	-0.057094	0.0887031	0.9991	
PCM,N	GDFL,N	0.0150387	0.0227835	-0.055687	0.0857640	0.9992	
GDFL,TB	C,TB	0.0146533	0.0233417	-0.057805	0.0871113	0.9995	
PCM,T	C,TB	0.0129504	0.0229368	-0.058251	0.0841516	0.9998	
GDFL,T	C,N	0.0123453	0.0228605	-0.058619	0.0833097	0.9998	
PCM,N	GDFL,T	0.0108740	0.0225665	-0.059178	0.0809257	0.9999	
GDFL,T	C,T	0.0104187	0.0227005	-0.060049	0.0808864	0.9999	
C,T	C,TB	0.0095506	0.0235944	-0.063692	0.0827931	1.0000	
GDFL,N	C,N	0.0081806	0.0230748	-0.063449	0.0798101	1.0000	
C,N	C,TB	0.0076240	0.0237484	-0.066096	0.0813445	1.0000	
GDFL,TB	C,N	0.0070293	0.0229303	-0.064152	0.0782104	1.0000	
GDFL,T	PCM,T	0.0070188	0.0220162	-0.061325	0.0753624	1.0000	
GDFL,N	C,T	0.0062540	0.0229162	-0.064883	0.0773914	1.0000	
PCM,T	C,N	0.0053265	0.0225181	-0.064575	0.0752278	1.0000	
GDFL,T	GDFL,TB	0.0053160	0.0224377	-0.064336	0.0749679	1.0000	
GDFL,TB	C,T	0.0051027	0.0227708	-0.065583	0.0757886	1.0000	
GDFL,T	GDFL,N	0.0041647	0.0225853	-0.065945	0.0742748	1.0000	
PCM,T	C,T	0.0033998	0.0223556	-0.065997	0.0727968	1.0000	
GDFL,N	PCM,T	0.0028542	0.0222386	-0.066180	0.0718880	1.0000	
C,T	C,N	0.0019266	0.0231875	-0.070053	0.0739061	.	
GDFL,TB	PCM,T	0.0017029	0.0220887	-0.066866	0.0702714	.	
GDFL,N	GDFL,TB	0.0011513	0.0226560	-0.069178	0.0714807	.	

Table 13: Ordered difference table for the mean severity of each scenario.

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
PCM,T	GDFL,T	1.150000	0.7607514	-1.24025	3.540249	0.8488	
PCM,T	C,TB	1.150000	0.7607514	-1.24025	3.540249	0.8488	
PCM,T	PCM,N	1.100000	0.7607514	-1.29025	3.490249	0.8782	
PCM,TB	GDFL,T	1.000000	0.7607514	-1.39025	3.390249	0.9258	
PCM,T	C,N	1.000000	0.7607514	-1.39025	3.390249	0.9258	
PCM,TB	C,TB	1.000000	0.7607514	-1.39025	3.390249	0.9258	
PCM,TB	PCM,N	0.950000	0.7607514	-1.44025	3.340249	0.9442	
PCM,TB	C,N	0.850000	0.7607514	-1.54025	3.240249	0.9709	
PCM,T	GDFL,N	0.800000	0.7607514	-1.59025	3.190249	0.9800	
PCM,T	C,T	0.700000	0.7607514	-1.69025	3.090249	0.9916	

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
PCM,TB	GDFL,N	0.650000	0.7607514	-1.74025	3.040249	0.9949	
PCM,T	GDFL,TB	0.600000	0.7607514	-1.79025	2.990249	0.9971	
GDFL,TB	GDFL,T	0.550000	0.7607514	-1.84025	2.940249	0.9984	
GDFL,TB	C,TB	0.550000	0.7607514	-1.84025	2.940249	0.9984	
PCM,TB	C,T	0.550000	0.7607514	-1.84025	2.940249	0.9984	
GDFL,TB	PCM,N	0.500000	0.7607514	-1.89025	2.890249	0.9992	
C,T	GDFL,T	0.450000	0.7607514	-1.94025	2.840249	0.9996	
C,T	C,TB	0.450000	0.7607514	-1.94025	2.840249	0.9996	
PCM,TB	GDFL,TB	0.450000	0.7607514	-1.94025	2.840249	0.9996	
C,T	PCM,N	0.400000	0.7607514	-1.99025	2.790249	0.9998	
GDFL,TB	C,N	0.400000	0.7607514	-1.99025	2.790249	0.9998	
GDFL,N	GDFL,T	0.350000	0.7607514	-2.04025	2.740249	0.9999	
GDFL,N	C,TB	0.350000	0.7607514	-2.04025	2.740249	0.9999	
GDFL,N	PCM,N	0.300000	0.7607514	-2.09025	2.690249	1.0000	
C,T	C,N	0.300000	0.7607514	-2.09025	2.690249	1.0000	
GDFL,N	C,N	0.200000	0.7607514	-2.19025	2.590249	1.0000	
GDFL,TB	GDFL,N	0.200000	0.7607514	-2.19025	2.590249	1.0000	
C,N	GDFL,T	0.150000	0.7607514	-2.24025	2.540249	1.0000	
C,N	C,TB	0.150000	0.7607514	-2.24025	2.540249	1.0000	
PCM,T	PCM,TB	0.150000	0.7607514	-2.24025	2.540249	1.0000	
C,N	PCM,N	0.100000	0.7607514	-2.29025	2.490249	1.0000	
C,T	GDFL,N	0.100000	0.7607514	-2.29025	2.490249	1.0000	
GDFL,TB	C,T	0.100000	0.7607514	-2.29025	2.490249	1.0000	
PCM,N	GDFL,T	0.050000	0.7607514	-2.34025	2.440249	.	
PCM,N	C,TB	0.050000	0.7607514	-2.34025	2.440249	.	
C,TB	GDFL,T	1.776e-15	0.7607514	-2.39025	2.390249	.	

Table 14: Ordered difference table comparing average number of fires for all scenarios.

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
GDFL,N	PCM,T	0.1288256	0.0868922	-0.140908	0.3985593	0.8639	
C,T	PCM,T	0.1264913	0.0873492	-0.144661	0.3976439	0.8789	
PCM,N	PCM,T	0.1119735	0.0868175	-0.157528	0.3814754	0.9345	
C,TB	PCM,T	0.1100012	0.0896201	-0.168201	0.3882031	0.9506	
PCM,TB	PCM,T	0.1070996	0.0866694	-0.161943	0.3761419	0.9486	
GDFL,N	GDFL,TB	0.1069641	0.0885229	-0.167832	0.3817599	0.9549	
C,T	GDFL,TB	0.1046298	0.0889716	-0.171559	0.3808185	0.9615	
PCM,N	GDFL,TB	0.0901119	0.0884496	-0.184456	0.3646802	0.9842	
C,TB	GDFL,TB	0.0881397	0.0912021	-0.194973	0.3712523	0.9888	
PCM,TB	GDFL,TB	0.0852380	0.0883043	-0.188879	0.3593552	0.9889	

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
GDFL,N	GDFL,T	0.0843082	0.0882468	-0.189631	0.3582471	0.9896	
C,T	GDFL,T	0.0819740	0.0886969	-0.193362	0.3573100	0.9916	
PCM,N	GDFL,T	0.0674561	0.0881733	-0.206255	0.3411667	0.9977	
C,N	PCM,T	0.0668233	0.0879840	-0.206300	0.3399464	0.9978	
C,TB	GDFL,T	0.0654839	0.0909341	-0.216797	0.3477648	0.9985	
PCM,TB	GDFL,T	0.0625822	0.0880275	-0.210676	0.3358403	0.9987	
GDFL,N	C,N	0.0620024	0.0901592	-0.217873	0.3418777	0.9989	
C,T	C,N	0.0596681	0.0905998	-0.221575	0.3409112	0.9992	
PCM,N	C,N	0.0451502	0.0900872	-0.234502	0.3248022	0.9999	
C,N	GDFL,TB	0.0449617	0.0895949	-0.233162	0.3230852	0.9999	
GDFL,T	PCM,T	0.0445174	0.0860233	-0.222519	0.3115539	0.9999	
C,TB	C,N	0.0431780	0.0927912	-0.244868	0.3312236	0.9999	
PCM,TB	C,N	0.0402763	0.0899446	-0.238933	0.3194854	1.0000	
GDFL,T	GDFL,TB	0.0226558	0.0876702	-0.249493	0.2948046	1.0000	
C,N	GDFL,T	0.0223059	0.0893221	-0.254971	0.2995827	1.0000	
GDFL,TB	PCM,T	0.0218616	0.0863065	-0.246054	0.2897771	1.0000	
GDFL,N	PCM,TB	0.0217260	0.0888768	-0.254168	0.2976205	1.0000	
C,T	PCM,TB	0.0193918	0.0893237	-0.257890	0.2966735	1.0000	
GDFL,N	C,TB	0.0188244	0.0917565	-0.266009	0.3036582	1.0000	
GDFL,N	PCM,N	0.0168521	0.0890212	-0.259491	0.2931948	1.0000	
C,T	C,TB	0.0164901	0.0921895	-0.269688	0.3026679	1.0000	
C,T	PCM,N	0.0145179	0.0894674	-0.263210	0.2922457	1.0000	
PCM,N	PCM,TB	0.0048739	0.0888038	-0.270794	0.2805417	.	
C,TB	PCM,TB	0.0029016	0.0915456	-0.281277	0.2870808	.	
GDFL,N	C,T	0.0023343	0.0895398	-0.275618	0.2802870	.	
PCM,N	C,TB	0.0019722	0.0916858	-0.282642	0.2865866	.	

Table 15: Ordered Difference tables comparing average fire size for all scenarios.

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
GDFL,T	PCM,N	0.3176045	0.1552523	-0.170192	0.8054009	0.5139	
C,N	PCM,N	0.2482755	0.1552523	-0.239521	0.7360718	0.8041	
GDFL,N	PCM,N	0.2397058	0.1552523	-0.248090	0.7275022	0.8330	
GDFL,TB	PCM,N	0.2076913	0.1552523	-0.280105	0.6954876	0.9185	
GDFL,T	PCM,TB	0.1977505	0.1552523	-0.290046	0.6855468	0.9376	
C,TB	PCM,N	0.1932800	0.1552523	-0.294516	0.6810763	0.9451	
GDFL,T	C,T	0.1877374	0.1552523	-0.300059	0.6755337	0.9535	
PCM,T	PCM,N	0.1818696	0.1552523	-0.305927	0.6696659	0.9614	
GDFL,T	PCM,T	0.1357349	0.1552523	-0.352061	0.6235313	0.9940	
C,T	PCM,N	0.1298671	0.1552523	-0.357929	0.6176634	0.9956	
C,N	PCM,TB	0.1284214	0.1552523	-0.359375	0.6162177	0.9959	
GDFL,T	C,TB	0.1243245	0.1552523	-0.363472	0.6121209	0.9967	
PCM,TB	PCM,N	0.1198541	0.1552523	-0.367942	0.6076504	0.9975	
GDFL,N	PCM,TB	0.1198518	0.1552523	-0.367945	0.6076481	0.9975	

Level	- Level	Difference	Std Err Dif	Lower CL	Upper CL	p-Value	Difference
C,N	C,T	0.1184084	0.1552523	-0.369388	0.6062047	0.9977	
GDFL,T	GDFL,TB	0.1099132	0.1552523	-0.377883	0.5977096	0.9986	
GDFL,N	C,T	0.1098387	0.1552523	-0.377958	0.5976350	0.9986	
GDFL,TB	PCM,TB	0.0878372	0.1552523	-0.399959	0.5756335	0.9997	
GDFL,T	GDFL,N	0.0778987	0.1552523	-0.409898	0.5656950	0.9999	
GDFL,TB	C,T	0.0778242	0.1552523	-0.409972	0.5656205	0.9999	
C,TB	PCM,TB	0.0734259	0.1552523	-0.414370	0.5612222	0.9999	
GDFL,T	C,N	0.0693291	0.1552523	-0.418467	0.5571254	1.0000	
C,N	PCM,T	0.0664059	0.1552523	-0.421390	0.5542022	1.0000	
C,TB	C,T	0.0634129	0.1552523	-0.424383	0.5512092	1.0000	
PCM,T	PCM,TB	0.0620155	0.1552523	-0.425781	0.5498118	1.0000	
GDFL,N	PCM,T	0.0578362	0.1552523	-0.429960	0.5456326	1.0000	
C,N	C,TB	0.0549955	0.1552523	-0.432801	0.5427918	1.0000	
PCM,T	C,T	0.0520025	0.1552523	-0.435794	0.5397988	1.0000	
GDFL,N	C,TB	0.0464258	0.1552523	-0.441370	0.5342222	1.0000	
C,N	GDFL,TB	0.0405842	0.1552523	-0.447212	0.5283805	1.0000	
GDFL,N	GDFL,TB	0.0320145	0.1552523	-0.455782	0.5198109	1.0000	
GDFL,TB	PCM,T	0.0258217	0.1552523	-0.461975	0.5136180	1.0000	
GDFL,TB	C,TB	0.0144113	0.1552523	-0.473385	0.5022076	1.0000	
C,TB	PCM,T	0.0114104	0.1552523	-0.476386	0.4992067	.	
C,T	PCM,TB	0.0100130	0.1552523	-0.477783	0.4978094	.	
C,N	GDFL,N	0.0085696	0.1552523	-0.479227	0.4963660	.	

Table 16: Ordered Difference table comparing FRI for all scenarios.

APPENDIX E: ADDITIONAL BACKGROUND ON COST ANALYSIS

A number of studies have evaluated how property values can be influenced by both proximity to wildfires and active fuel treatments. Such investigations are particularly pertinent to Tejon as plans are in place for extensive developments. Loomis (2004) analyzed real estate data from the town of Pine, Colorado for three years before and five years after a serious wildfire burned through the area and destroyed 10 houses. His results indicated that property values in the wake of the fire dropped by 15%. Despite the benefits associated with living in a rural landscape, Loomis found that recent fire events caused declining property values even on lots that showed no visible sign of fire damage. Aside from perceptions of fire danger, it is argued that these wildfires detract from the amenity values of an area, and leave behind a charred landscape with reduced recreational opportunities and burned out trails and roads (Loomis, 2004).

Kim and Wells (2005) analyzed how property values change with particular fuel management actions by gathering property value data from real estate agencies and then examining satellite images of each property to determine the associated forest density. Lower forest density was due to owners actively thinning out vegetation and fuels on their property. After standardizing property values based on variables such as house size and location, they found that lower density forest cover around homes was indeed correlated with a higher property value. For forest cover that had undergone intense mechanical thinning, followed by removal of all cut and downed wood, the average increase was more than $40,000. The typical cost of mechanically thinning a half-kilometer area around a home was about $30,000, indicating that the net present value (NPV) associated with forest thinning around private property provides about $10,000 in property value benefits. The authors acknowledge that part of this benefit could accrue from enhanced scenery associated with thinning, but posit that a large part of the value is from homeowners' perceptions of fire safety related to decreased density (Kim & Wells, 2005).

APPENDIX F: BACKGROUND ON ADDITIONAL WILDLIFE SPECIES AFFECTED BY FIRE

1. Burrowing Owl

Knowledge regarding the effects of fire on burrowing owls (*Athene cunicularia hypugaea*) is limited, though it has been suggested that they may benefit from vegetation height reductions (Reiner, 2007). Supporting this hypothesis is the fact that burrowing owls breed in areas with limited plant cover. It is argued that limited plant cover may increase visibility and create habitat conditions that support prey populations (Green & Anthony, 1989). Furthermore, the most suitable habitats for burrowing owls, as rated by the California Wildlife Habitat Relationships (CWHR) Database (2008), include alkali desert scrub, annual and perennial grassland, desert scrub, desert succulent shrub, Joshua tree woodland, and low sage with limited vegetation cover. Burrowing owls have also been documented moving into areas after a fire has occurred (Green & Anthony, 1989).

2. California Mule Deer

California mule deer (*Odocoileus hemionus californicus*) favor chaparral and oak woodland habitats for feeding, cover, and reproduction (Clark, 2004; Bowyer, 1986; California Wildlife Habitat Relationships System, 2008). Enhanced forage value and access for mule deer is sometimes an objective of prescribed burning in chaparral (Shaffer & Laudenslayer, 2006; Dasmann & Dasmann, 1963). But Klinger et al. (1989) found that where chaparral is within close proximity to other suitable habitat types, such as oak woodlands or grasslands, fire does not lead to an increase deer survival. Deer populations increased in burned chaparral only after the second growing season and then declined to pre-burn numbers within six months (Klinger et al., 1989). Thus, prescribed burns in the Ranch's chaparral will not necessarily benefit California mule deer.

3. California Red-Legged Frog

The endangered California red-legged frog (*Rana draytonii*) has not been found on the Ranch, but is likely to be present on Tejon, El Paso, and Tunis Creeks (Arcata Fish and Wildlife Office, 2011; Jennings & Hartesveldt, 2011). The recovery plan for the frog explicitly discusses fire management (Arcata Fish and Wildlife Office, 2011). Among the suggestions are avoiding emergency fire suppression activities near frog habitat. This includes placing staging areas and emergency water sources away from breeding pools. The recovery plan also recommends using prescribed burning to improve the habitat and decrease the chance of severe fire, but only in seasons when frogs are not dispersing (U.S. Fish and Wildlife Service, 2002).

4. Desert Tortoise

A recent survey of reptiles and amphibians in the Acquisition Areas found a "high likelihood" that the federally-listed desert tortoise (*Gopherus agassizii*) is present on the Ranch (Jennings & Hartesveldt, 2011). If present, it could be significantly affected by fire management decisions in desert scrub environments (Esque et al., 2003). Decisions that lead to increased wildfire may harm it both directly and indirectly, killing individual tortoises and changing vegetation cover in ways that make survivors more vulnerable to predation and extreme temperatures (Esque et al., 2003). Conversely, management decisions that decrease the frequency and severity of wildfire may benefit desert tortoises on the Ranch now, and allow the Ranch to serve as a refuge for this species in the future. This is particularly important in desert scrub and desert wash, which the CWHR Database rates as highly suitable habitat for reproduction, cover and feeding (Marlow, 2000). But it is also a management consideration in desert succulent shrub and Joshua tree woodlands, which CWHR rates as moderately suitable habitat (Marlow, 2000).

5. Purple Martin

The purple martin (*Progne subis*) is a rare summer resident of oak woodlands, conifer forests, and riparian areas throughout California (Dudek, 2009a). Purple martins are cavity nesters, constructing nests in abandoned woodpecker holes or other existing cavities in both live trees and dead snags (Dudek, 2009a; Williams, 2002). Fire can have variable effects on cavity-nesters: moderate to high severity wildfires often create new snags, but prescribed burns and low severity wildfires may consume existing snags while creating relatively few new ones (Bagne et al., 2008). In the Tehachapi Mountains, the majority of purple martins appear to nest in living oak trees rather than dead snags, and thus this population may not be dependent on fire for the creation of nesting sites (Williams, 2002; Airola & Williams, 2008). Purple martins on Tejon Ranch have primarily been observed nesting in large valley oak trees (Tejon Ranch Conservancy, 2011). Thus, in the long-term, purple martins will likely benefit from management practices that increase the extent of valley oak woodlands and savannahs on the Ranch.

6. Tehachapi Pocket Mouse

The Tehachapi pocket mouse (*Perognathus alticolus inexpectatus*), a federal Species of Concern, has been found in several of the Ranch's arid communities, including chaparral, Joshua tree woodlands, and California buckwheat-dominated grasslands (Cypher et al., 2010). It is believed to occur throughout the Bi-Centennial and Tri-Centennial acquisition areas, which may represent a significant portion of its overall range (Cypher et al., 2010). Because of its geographic concentration, it could be seriously affected by a major wildfire in these areas, or by type conversion. Given its use of burrows for cover and reproduction, it could also be affected by changes in the consistency of soil, including the erosion and loss of structure sometimes associated with wildfire (Brylski, 2008).

7. Tehachapi Slender Salamander

The Tehachapi slender salamander (*Batrachoseps stebbinsi*) is found in shaded canyons on north-facing slopes between 2,500 and 8,300 feet in elevation (Dudek, 2009a; CaliforniaHerps, 2012; U.S. Fish and Wildlife Service, 2011a). The species is confined to habitats where canyon live oak and rock talus are present (Dudek, 2009a). Tehachapi slender salamanders are rare and sedentary, which makes them highly vulnerable to extirpation due to large wildfires or other disturbances. They typically remain underground during the summer dry season when most fires occur, which may mitigate direct impacts of fire. But burned areas may remain unsuitable for habitation for several years due to decreased surface moisture and understory cover (U.S. Fish and Wildlife Service, 2011a). Over the long term, Tehachapi slender salamanders may benefit from patterns of wildfire that maintain canyon live oak as a component of the overstory.

8. Willow Flycatcher

The Southwestern willow flycatcher (*Empidonax traillii*), a federally endangered species threatened by fragmentation and degradation of habitat, has been found on the Ranch (Craig & Williams, 1998). During its May-August breeding season, the flycatcher nests in riparian areas dominated by willows and cottonwoods, as well as moist meadows. For this reason, any fuel management should be conducted outside of willow flycatcher mating season. Tamarisk removal, while improving other aspects of the riparian zone, may reduce willow flycatcher habitat.