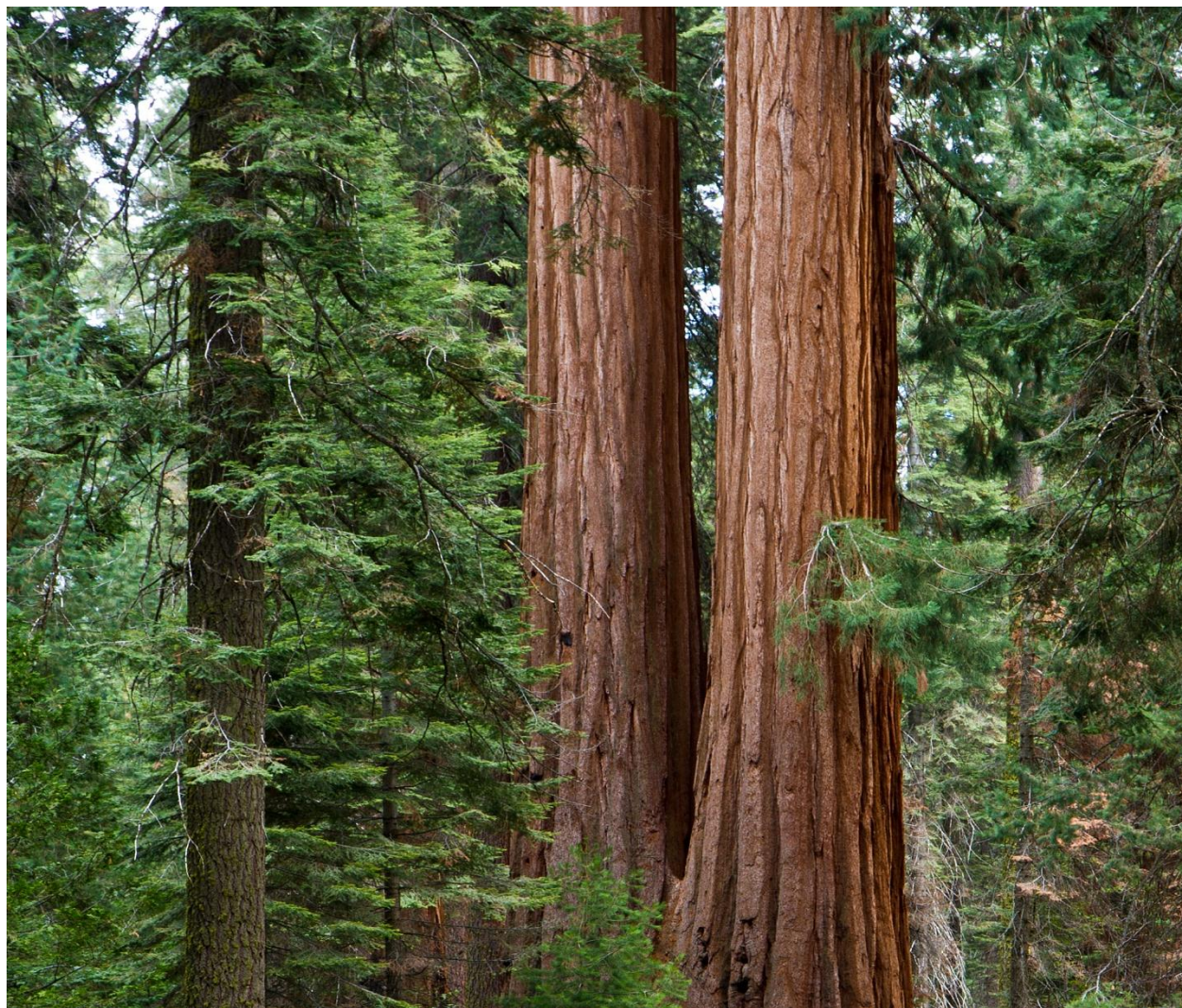




A Natural Resource Condition Assessment for Sequoia and Kings Canyon National Parks

Appendix 10 - Foothills Vegetation

Natural Resource Report NPS/SEKI/ NRR—2013/665.10



ON THE COVER

Giant Forest, Sequoia National Park
Photography by: Brent Paull

A Natural Resource Condition Assessment for Sequoia and Kings Canyon National Parks

Appendix 10 - Foothills Vegetation

Natural Resource Report NPS/SEKI/ NRR—2013/665.10

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Fort Collins, Colorado

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Scope of Analysis

The foothill zone in the Sequoia and Kings Canyon National Parks (SEKI) encompasses areas between 460 and 1830 meters above sea level (m.a.s.l.) along the watershed of the Kaweah and Kings rivers (Parsons et al. 1981). Foothill vegetation is typically oak woodlands, hardwood forests, and shrublands. Hardwood forests cover most of the foothills (60% of the total area), most of which (42%) corresponds to evergreen forest dominated by canyon live oak (*Quercus chrysolepsi*) associations. Shrublands, the second most abundant cover type (27%), are particularly characterized by chamise (*Adenostoma fasciculatum*)-chaparral associations (17%). Mixed coniferous forests are minimally represented at this elevation range (9%) (Table 1).

Foothill plant communities are influenced by several potential stressors that may affect their stability and conservation status. In SEKI as a whole, these potential stressors include air pollution, climate change, altered fire regimes, invasive species, and infectious diseases. In the particular case of the foothills, some of these stressors represent a more immediate threat. In the foothill shrublands, for instance, more frequent fire regimes favor invasion of non-native annual grasses and limit canopy closure of the native vegetation (Keeley et al. 2003). We also add cattle grazing to the list of potential stressors in foothill communities. While still a subject of some debate, it is generally thought that frequent fire and increased non-native grasses, in combination with severe climatic events and cattle grazing, can interfere with regeneration of oak species in foothill oak woodlands.

Of all the foothill vegetation types at SEKI, we pay particular attention to blue oak (*Quercus douglasii*) woodlands. Blue oak woodlands are unique and valuable areas in California (Standiford et al. 2001). Over the past several decades, the future vitality and existence of blue oaks have become the focus of extensive research. Blue oak woodlands are anticipated to diminish further in the coming years given that several oak species, including blue oaks, appear to be regenerating at levels below what is necessary for self-replacement.

In this chapter, we evaluated existing information on factors affecting persistence of two different vegetation types in the foothills—hardwood forests and shrublands—and specifically evaluated the current status of blue oak populations in the park. To do this, we compiled and evaluated vegetation information pertinent to the foothills using five available datasets. Each dataset was generated as part of independent studies conducted throughout SEKI between 1960 and 2004, and they cover most of the foothill area. To focus on the conservation status of oak woodlands in particular, we conducted a more in-depth literature review on the current state of knowledge of blue oak populations in the park.

Critical Questions

We had two objectives. First, we characterized the status of foothill vegetation and explored if spatial variability in community-level metrics such as diversity can be associated with potential stressors in the foothill zone (specifically, cattle grazing, fire and invasive species). We focused this analysis on overall community characteristics for the two most abundant vegetation types in the foothills: hardwood forests, which include blue oak woodlands and oak forests, and shrublands. Within each vegetation type, we estimated diversity and dominance for native herbaceous species, shrubs, and tree species. We focused on the spatial variation of community characteristics, emphasizing the relationships with spatial variation of topographic and climatic variability, history of fire and cattle grazing.

Second, we performed a comprehensive literature review on the factors affecting blue oak regeneration throughout California, and in light of this review, we evaluated the general condition of blue oak woodlands in SEKI. We focused on blue oak because it is the most extensive hardwood species in the state, encompassing an estimated 1.214 million ha (3 million acres) and representing nearly half of the oak-covered landscape in California (Bolsinger 1988; Allen-Diaz and Holzman 1991). In addition, blue oak has served as a case study species for over 60 years in studies on the low regeneration observed in oak communities; it is one of the most well studied species of oak.

Given these two objectives, we investigated three critical questions:

1. What is the status of native herbaceous, shrub, and tree communities throughout the foothill zone?
2. To what extent is the spatial variation of diversity metrics in the foothills explained by variability in local growing conditions, characterized by topography and climatic variation, and to what extent is it explained by past history of disturbance, mainly fire and grazing?
3. In the light of our current knowledge of factors affecting blue oak regeneration, what is the status of blue oak communities at SEKI?

Table 1. Vegetation types within foothills areas in each HUC-10 unit. The percentage of the foothill area of SEKI NPS land in each HUC-10 unit is listed in parentheses, followed by acronyms used for each unit (e.g., MFK for Middle Fork of Kings River); these acronyms are used throughout this chapter. Except for Middle and Marble Fork Kaweah units the HUC-10 delineation corresponds to watershed units; for this reason we refer to HUC-10 units rather than watersheds. Percentages do not add to 100% in each column due to rounding.

HUC 10 unit classification	Roaring River (0.13%)	Middle Fork Kings (4.8%) MFK	Lower South Fork Kings (3.2%) LSFk	Upper South Fork Kings (2.5%) USFK	Mill Flat Creek (0.03%)	Dye Creek (0.03%)	Marble Fork Kaweah (14.3%) MbFKw	North Fork Kaweah (26%) NFKw	Middle Fork Kaweah (20%) MFKw	East Fork Kaweah (23%) EFKw	South Fork Kaweah (6.2%) SFKw	North Fork Tule (0.01%) NFT	GRAND TOTAL
Deciduous shrubland		4.9%	1.4%	2.3%	7.4%	10%	3.1%	7.1%	12%	19%	13%		10%
Evergreen shrubland	1.6%	3.2%	0.75%	2.5%			37%	21%	12%	16%	0.49%		17%
Mixed shrub/herbaceous			0.21%		11%		0.07%		0.16%	0.50%			0.17%
Deciduous forest	5.9%	2.1%	3.3%	9.7%			0.17%	0.85%	0.73%	0.69%	0.59%		1.2%
Deciduous woodland							12%	7.8%	6.6%	2.7%	3.3%		5.8%
Evergreen forest	45%	43%	37%	45%	33%	66%	36%	41%	46%	41%	48%		42%
Mixed evergreen deciduous forest	0.43%	3.1%	3.3%	7.3%	47%	24%	4.8%	13%	14%	10%	23%	100%	11%
Barren	37%	1.9%	24%	27%	1.6%		2.7%	1.9%	2.5%	2.1%	2.1%		4.0%
Mixed Coniferous Forest	9.2%	41%	30%	6.9%	0.96%		4.8%	8.4%	5.8%	8.8%	9.6%		9.1%

Data Sources and Types Used in Analysis

Vegetation datasets.

In our analysis, we used five different datasets from independent studies conducted in the park since 1969 (Table 2; Figure 1). Out of the five available datasets, the vegetation mapping plots for accuracy assessment (VMA) are by far the most numerous and widely distributed throughout the foothills ($n=389$). These plots were specifically designed to support an ongoing mapping effort and were surveyed between 2002 and 2004, and their location is biased towards easily accessible areas. The second most abundant set, the Nature Resource Inventory (NRI) plots ($n=77$), were selected systematically through randomized sampling, which provided us with the most reliable description of the sampled areas. The low number of plots of this data set within the foothills limited the precision of our analyses because plots do not cover the full range of environments of the foothills (Figure 1). Despite this caveat, we use NRI plots over VMA plots to explore spatial patterns across the foothills due to their unbiased sampling scheme. For this reason, we can only explore spatial patterns in some foothill areas in the park; for instance, we do not have enough information to evaluate south, and southeast areas of Sequoia National Park.

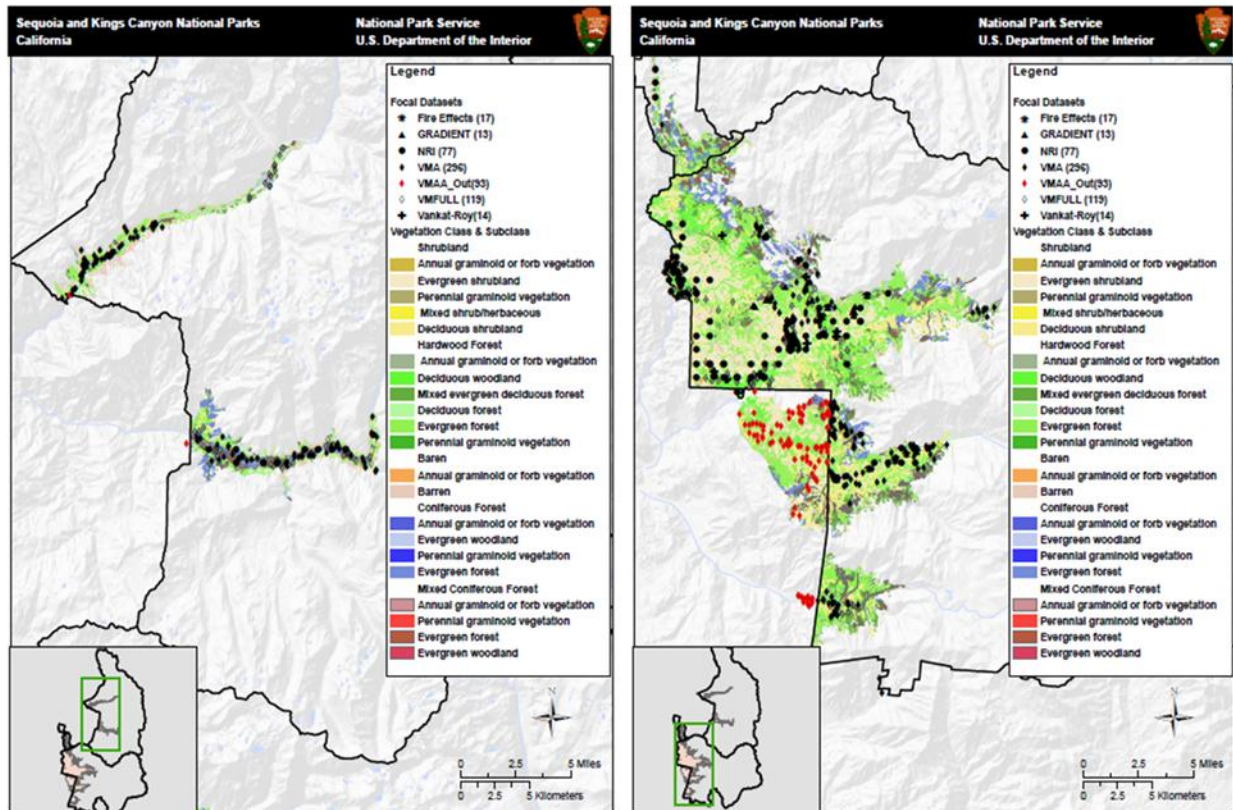


Figure 1. Location of plots used for the analysis of condition of plant communities at SEKI. Plots from different studies are indicated with different symbols. Left panel corresponds to Kings Canyon National Park and right panel to Sequoia National Park. We largely focus on spatial variation related to NRI plots (black circles). Note that some of the VMA plots occur outside of National Park boundaries (red diamonds); because they were outside of the park, they were not considered in analyses with stressors except for explicit consideration in grazing analyses. Abbreviations: NRI (Natural resource inventory plots), VMA (Vegetation Mapping plots for Accuracy assessment), VMAA_out (Vegetation Mapping plots

Table 2. Summary of available datasets for the analysis of current conditions in the foothill zone. In our analyses, we concentrate on NRI plots, as they were selected with a stratified randomized sampling scheme and encompassed a wide range of foothill plant communities. VMA plots, while more numerous than NRI plots, were more biased in location; we used these as a secondary data source for spatial analyses. The remaining three datasets were collected with specific purposes and have low sample sizes (n=13 to 17) in the foothills; we reserve the use of these datasets for focused analyses.

Data Set	Purpose	Year Sampled	Sampling Design	Sampled Area	Data recorded	Number of plots	Range Elevation (m a.s.l.)
Natural Resource Inventory Plots (NRI)	Describe the distribution and abundance of vascular plants	1985-1996	Stratified random sampling scheme using the 1-km UTM coordinate grid	0.1 ha circular plots	-DBH trees>1.3m -Herb Cover/Species Cover/Species -	77	475-1830
Vegetation Mapping Plots for accuracy assessment (VMA)	Support mapping efforts throughout the park	2000-2004	Stratified by vegetation map unit and location. Heavy sampling in accessible locations close to roads and creeks	0.1 ha circular plots	Sapling/Species - Diameter Classes of Trees >1.4 m -Herb Cover/Species -Shrub Cover/Species	389 (+ 93 outside SEKI)	396-1830
Full Vegetation Mapping plots (VM Full)	Same as VMA	2000-2004	Same as VMA	Same as VMA	-Diameter Classes of Trees >1.4 m	119	464-1830
Gradient Plots (GP)	Describe the distribution of forest trees along environmental gradients	1983	Chosen to cover as wide a range as possible of environmental conditions	0.1 ha rectangular plots	-Diameter Classes of Trees >1.4 m -Herb Cover/Species -Shrub Cover/Species	13	1341-1826
Vankat-Roy Plots (VR)	Monitor long-term change in tree communities	1966 and 1996	On representative vegetation types along an elevation gradient	0.01 ha strips	-Diameter Classes of Trees and shrubs with DBH >1.3 m	14	354-670
Fire Effect Plots (FF)	Monitor pre- and post-fire fuel and vegetation conditions	1988-2009	Random selection within vegetation type	-In shrubland areas 30 m strips -In forest 0.1 ha plots	-DBH of all trees -Herb Cover/Species -Shrub Cover/Species	17	743-1793

To compare the spatial variation of community metrics we used the HUC-10 digit classification (established by the California Interagency Watershed Mapping Committee-IWMC), which roughly corresponds to watershed units used by SEKI (Table 1). It is important to note that HUC-10 units do not exactly match the delineation for Marble and Middle Fork Kaweah watersheds, and thus we will use the term HUC-10 unit (or unit with the specific name is given) instead of watershed. We did not have enough information from the NRI dataset to evaluate community patterns along Roaring River, South Fork Kaweah, North Fork Tule, Mill Flat Creek, and Dye Creek units.

In addition to these two datasets, we used three other datasets. We used plots called the fire effects plots (FE), to explore community response to fire events. These plots are exclusively located on the North, Middle and East Fork Kaweah River units. Two other datasets, the Gradient and Vankat-Roy plots, were infrequent and did not allow an extensive characterization of most foothill areas since they were used to describe more specific and spatially restrictive questions. Vankat-Roy plots were designed to characterize specific areas in the park in 1969 and were later resampled in 1996 with the purpose of assessing fire effects and general vegetation changes. Given that more extensive analyses have been previously published (Roy and Vankat 1999; Vankat 1970; Vankat and Major 1978), we only present summary metrics for these plots.

Climate and topographic variables.

We focused on a series of topographic and climatic variables thought to be important in structuring foothill vegetation communities (Table 3, with values summarized in Table D of the appendix). These include topographic characteristics (elevation, aspect, slope), temperature (divided into growing and non-growing season), precipitation (also divided into growing and non-growing season) and landscape characteristics (distance from roads and streams).

Potential stressors.

We concentrate our analyses on three potential stressors (invasive grasses, cattle grazing and fire) thought to be important in foothills vegetation. Grazing information was obtained directly from SEKI personnel and consisted of two variables. The first variable identified areas in the foothills that experienced unauthorized grazing up to 1986, when some of these areas were fenced, which significantly reduced grazing until 1997. In 1997 more fences were put in place to control trespassing (Harold Werner, pers. communication). In addition to this variable, we also identified areas in the foothills that are designated for authorized stock grazing (mainly mule and horses). Since we do not have detailed information on the intensity and frequency of use in both of these areas, our analysis is limited to comparisons between areas where grazing had been detected and comparable areas in elevation where no grazing had been recorded. According to this classification, areas that had experienced grazing are restricted to the west border of the park in the North Fork Kaweah River and the Marble Fork Kaweah River units (Figure 2). In particular, authorized grazing has exclusively occurred on the southernmost border of the Marble Fork Kaweah unit, which is around the area where the park headquarters are located.

Given the foothills have been actively protected from cattle grazing and logging, among other uses, we estimate the effect of this protection on biodiversity metrics. For this effect we calculated differences among VMA plots located right outside the park (red diamonds in Figure 1) boundaries along the East fork Kaweah and areas in the park at comparable elevations in the

same HUC-10 unit. We use the current (1940s) park boundaries for this analysis. It is important to note that large areas of SEKI were added to the original park in 1926 and most of King Canyon in 1940. Although after the expansions, existing grazing allotments continued to be grazed until the 1980s in both parks.

To describe fire regimes, we used the index of fire return interval departure (FRID) that categorizes areas where fire frequency has been as expected from historical records (high scores) and where fire frequency is severely far from the historical norm (low scores) (Table 5). These scores were designed to identify priority areas for fire management. Most areas in the foothills have high FRID scores (in 2009, 36% scored 2, 29% scored 1, 21% scored 3, and 8.9 scored 4, especially at medium and high elevations in Sequoia National Park. Although it is assumed that fire regimes probably changed at the end of the Little Ice Age and the end of Native American burning, very little is known about the historical fire regime at low elevations because fires are not clearly recorded in the annual growth rings of non-coniferous species. Given high level of uncertainty in historical fire regimes in the foothill that translates in high uncertainty in FRID values (Caprio and Lineback 2002), our analyses based on FRID should be considered a first approximation to relate community characteristics with departures from fire regimes.

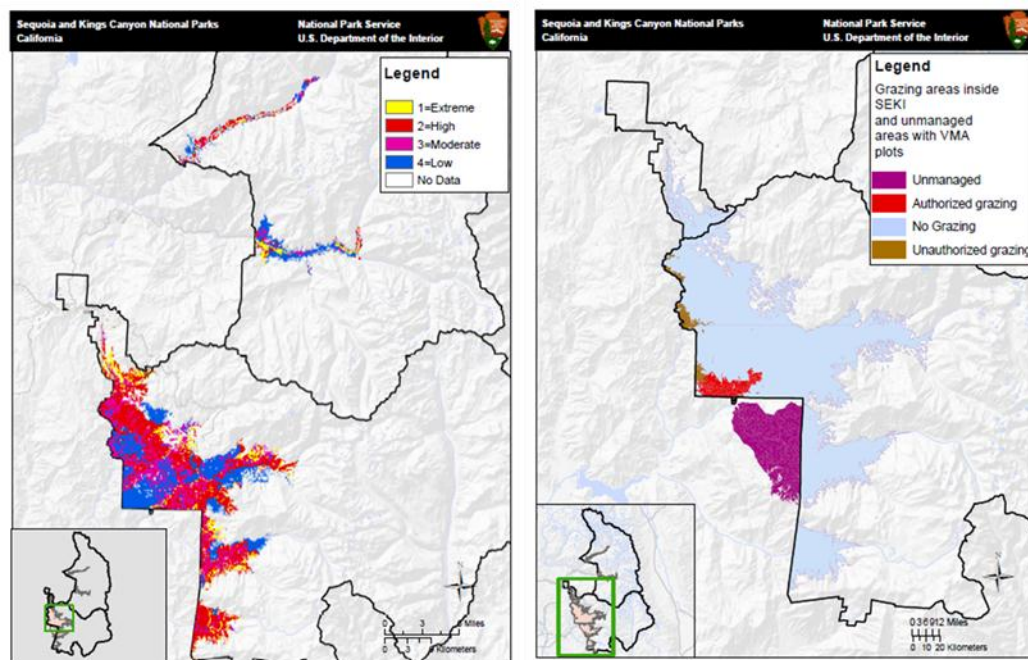


Figure 2. Values of two main stressors throughout the foothills. Right panel corresponds to FRID values and right panel to areas that have experienced or are still experiencing livestock grazing throughout the foothill zone of Sequoia National Park. Authorized grazed areas refer to areas of administrative use where stock pasture is authorized. Cattle grazed areas refer to areas that experienced unauthorized grazing until 1986, and that could have experienced light grazing up to 1997 when the park completed fences to prevent trespassing. Information for this variable was provided by SEKI. We used plots outside SEKI in the VMA data sets (unmanaged areas) to estimate the effect of park protection policies.

In addition, we explored the general disturbance history as described by a composite index showing the magnitude and duration of vegetation cover losses (regardless of cause) between 1985 and 2009 and obtained by Landsat image interpretation (R. Kennedy, Sierra Nevada Inventory and Monitoring Network, 2010) (Table 5). This index is based on reflectance values, and this not only identifies areas that have experienced fire, but also areas with significant tree or shrub mortality due to disturbance events or pathogens. In addition to duration and magnitude of disturbance, we also included a categorical variable that identified areas that have and have not experienced disturbance in the last 24 years (Figure 3). One of the limitations of this dataset is that it only provides information on the most prominent disturbances experienced in the last 24 years, and not on repeated disturbance events. It is also important to note that the NRI plots, on which we base our core analyses, were sampled between 1985 and 1996, and so this general disturbance descriptor should be interpreted as describing area having more or less probability of disturbance rather than the vegetation analyses describing response to particular disturbance events *per se*.

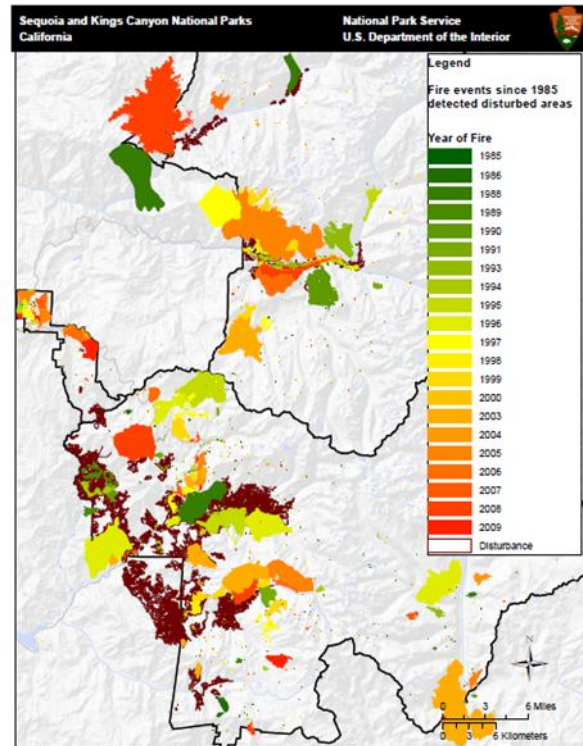


Figure 3. Areas that have experience fires since 1985 with respect to areas with detected vegetation cover loss. Areas with detected vegetation loss as reported by Kennedy (2010) are in dark red; all other polygons correspond to areas that have experienced fires in the last 26 years. Years of each fire are indicated in the legend with green colors representing early fires, and red colors more recent fires.

Based on this general disturbance index, changes in vegetation cover were detected throughout the foothills at SEKI. In general, disturbed areas were located at low elevations in all HUC-10 units, except on the South Fork Kaweah River unit, and on high elevations on the Middle, East, and South Fork Kaweah units. Several areas in the foothills experienced intense changes, declining in woody cover by almost 90%. These changes tended to last less than ten years. In most cases they do not last more than two years, and is stronger in hardwood forests ($R^2=0.43$, p -value <0.05) than in shrublands ($R^2=0.22$, p -value <0.05). Several of these vegetation changes correspond with major fires, especially in the Marble and Middle forks of the Kaweah River (Figure 3). Fires in the late 1980s and early 1990s affected the westernmost boundary of the North Fork Kaweah River, and high elevations at the Middle Fork. Fires in the late 1990s occurred close to the main park entrance, at lower elevations on the Marble Fork unit. Most recent fires have affected the Kings Canyon National Park, and the North and East Fork Kaweah River unit. Despite this coincidence, detected disturbance usually extended beyond the fire footprints, highlighting the potential importance of other causes vegetation loss (e.g. insect damage or fire independent tree mortality) in community dynamics.

Spatial and Temporal Analyses: Methods

General conditions and spatial patterns

Our general approach to explore spatial distribution in the foothill vegetation communities involved four steps (Figure 4). First, we estimated and summarized community level diversity metrics for each plot and explored general patterns at the HUC-10 unit level. For each plot we estimated the Shannon index of diversity ($H' = \sum_{i=1}^S (p_i \ln(p_i))$) where S is the total number of species considered and p_i relative abundance of species i , discriminating among native herbs, shrubs, and trees. During our analyses of the spatial variation of diversity, we also estimated spatial patterns of dominance using evenness; which derives from H' ($J' = \frac{H'}{\ln(S)}$). In addition, we calculated the percentage of native grasses, and abundance of the most common shrub and tree species in the database. We included the three most common shrub species in the foothills: chamise, birchleaf mountain mahogany (*Cercocarpus betuloides* Torr. & A. Gray), mountain misery (*Chamaebatia foliolosa* Benth.), and the three most common tree species: incense cedar (*Calocedrus decurrens* -Torr.- Florin), canyon live oak (*Quercus chrysolepis* Liebm.), and california black oak (*Quercus kelloggii* Newb). These metrics provide a general description of community composition by considering number of species, their relative abundance, and their overall dominance; in this sense they are preferable to just richness. We calculate these metrics for vegetation census plots in all five datasets; nevertheless, due to difference in species identification protocols between datasets, we focus on the variability within each dataset rather than comparisons across datasets.

Second, to describe spatial variation in these diversity metrics for NRI plots, we standardized diversity metrics using local Z- scores derived from hot-spot analysis in ArcGIS (Esri Inc., v.10). This score is calculated locally by summing the value of interest in a plot and its neighboring plots, and comparing it to the sum for all plots; significance of Z-scores is evaluated using standard normal distribution. Values higher than 1.96 or lower than -1.96 are considered significant deviations from the rest of the dataset with a 95% confidence level (i.e., these are areas with particularly high or low diversity for a given metric within each dataset).

Third, we explored the dependency of plot level metrics on topographic and climatic variables that characterize each plot, independent of the influence of potential stressors (Table 3). Our objective was to detect explanatory variables that are closely related with the intrinsic spatial variation of each metric. As many environmental variables are correlated with one another, we used a factor analyses (Johnson and Wichern 2007) to generate new composite variables as combinations of these topographic and climatic variables; we used five factors that explain almost 95% of the variability in the dataset (Table 4). Factor 1 explains more than half of the variability, and we interpreted it as describing the effects of elevation and climatic variables: when values in factor 1 increase, elevation and precipitation increase and temperature decreases. Factor 2 is solely associated with aspect, while factor 3 is associated with distance to streams, with high factor values representing areas away from streams. Contrary to factor 1, factor 4 values increase when elevation decreases and temperature increases. Finally factor 5 relates to slope and distance from roads; when areas are isolated and steep, values on this factor are high. We then used multiple regression models with spatial autocorrelated errors to explore the relationship of the Z-score for each diversity metric (explained above) and the composite factors derived from topographic and climatic variables.

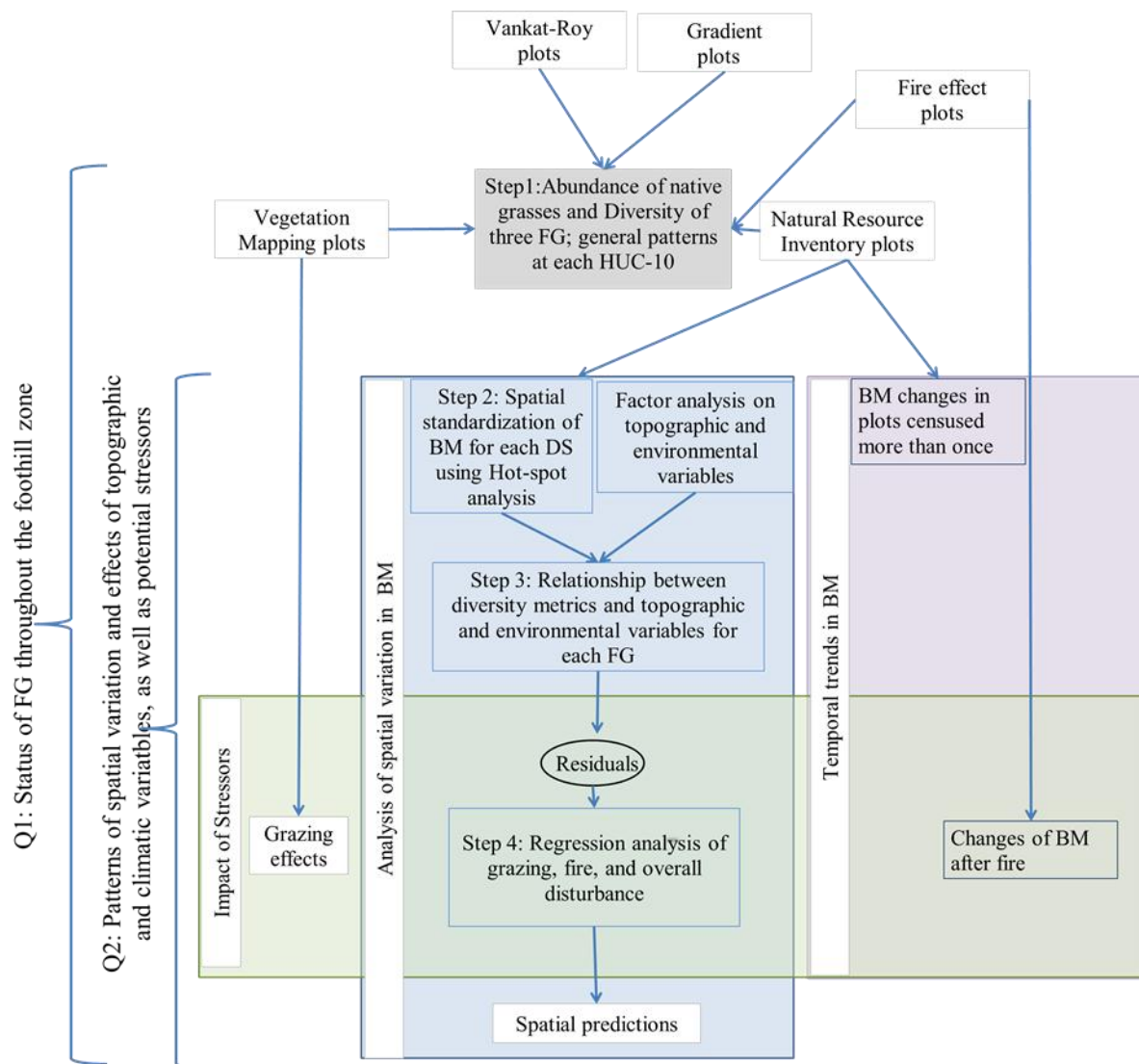


Figure 4. General framework for the analysis addressing the first two focal questions in this chapter. Four analytical steps and the datasets used are shown. Abbreviations: FG=Functional group, which are herbs, shrubs and trees; BM=Biodiversity metrics; DS=dataset, HUC-10=Hydric unit code, which is the same as the watershed.

Table 3. Topographic and climatic variables used to characterize each map unit. The source is indicated for each variable; we also indicate further manipulations for the datasets used for our analysis.

Type	Variable Name	Description	Source	Further manipulation
Topographic	Elevation (m.a.s.l)	Altitude in meters above the sea level	SEKI	None
	Aspect	Azimuth in degrees of slope surface	SEKI	Calculated from Elevation
	Slope	Percentage change of elevation over a specific area	SEKI	Calculated from Elevation
Environmental	Temperature during the growing season (°C)	Multiyear average of monthly temperature from 1971 to 2000	Bob Rice/PRISM Water quantity resource	Average, maximum and minimum between November and April
	Temperature during non-growing season months (°C)		Bob Rice/PRISM Water quantity resource	Average, maximum and minimum between May and October
	Precipitation during the growing season (mm)	Multiyear average of total monthly precipitation from 1971 to 2000	Bob Rice/PRISM Water quantity resource	Total, maximum and minimum between November and April
	Precipitation during non-growing season months (mm)		Bob Rice/PRISM Water quantity resource	Total, maximum and minimum between May and October

Table 4. Rotated loadings for each factor with respect to each of the original explanatory variables. Extreme loading values indicate a high correlation between a factor and the explanatory variable. The last two rows indicate the percentage variability explained by each factor, and the cumulative explained variability. Acronyms for variables used are as follow: MnGST = minimum growing season temperature, MxNGST = maximum non-growing season temperature, GSP = precipitation during growing season months, NGSP = precipitation during non-growing season months, Dist. Road = distance to main roads in meters, and Dist. Stream = distance to closest stream in meters.

	Factor 1	Factor 2	Factor 3	Factor 4	Factor 5
Elev	0.897	-0.026	0.117	-0.267	0.081
Slope	-0.022	-0.018	0.08	-0.029	0.365
Aspect	-0.017	0.999	0.036	-0.018	-0.031
MnGST	-0.863	0.034	0.041	0.433	0.198
MxNGST	-0.895	0.009	-0.014	0.382	-0.072
GSP	0.958	0.014	-0.002	0.005	0.027
NGSP	0.956	0.005	0.01	0.071	0.03
Dist Roads	0.28	0.041	-0.031	-0.391	0.319
Dist Stream	0.047	0.039	0.982	0.018	0.18
Percent Explained	51.226	13.319	11.356	10.256	8.389
Cumulative Percent Explained	51.226	64.546	75.901	86.158	94.546

Finally, we investigated the influence of two potential stressors (cattle grazing and fire) as well as a composite index describing overall disturbance patterns detected by remote sensing (regardless of cause) (Table 5). To account for the spatial pattern in climate and topographical variables, we used residuals (the unexplained variance) from the multiple regression analysis.

Thus, this approach describes the influence of potential stressors on diversity after climate and topographic variation has been taken into account. In cases where environmental variables and potential stressors are tightly correlated, it is statistically impossible to distinguish which variable directly influences diversity: our approach here is conservative in terms of the influence of potential stressors as it only describes the influence of these potential stressors after the variability in environmental characteristics have been accounted for.

Table 5. Summary of variables depicting the effect of potential stressors of (fire, grazing history, and general disturbance characteristics) throughout the foothill zone in SEKI.

Category	General description	Variable	Description	Scale	Source
Fire effects	Fire return interval departures (FRID)	Fire Return Interval Departure	The severity of fire return interval departures considering the historical fire regime. Highly uncertain for low elevation foothill areas.	1=extreme departure 2=High,3=moderate, 4=low, 99=unburnable substrate	SEKI
		Trespassing	Evidence of trespassing cattle grazing in 1986	Categorical	Werner, 1986
Grazing	Cattle and Stock Grazing..	Authorized pastured areas	Areas authorized for stock grazing (mule and horses)	Categorical	SEKI
		Outside	Comparison of VMA plots inside and outside current park boundaries, comparable elevation range	Categorical	VMA dataset
Disturbance	Landsat monitoring of vegetation cover changes	Disturbance Magnitude	Relative change in percent vegetation cover after disturbance for all vegetation cover types	1-100%, 100% indicates complete loss of cover	Kennedy (2010)
		Disturbance Duration	Measures the duration of the effect detected	Years	Kennedy (2010)
		Disturbance Year			Kennedy (2010)
		Disturbance Category	Categorized areas as disturbed or not in the year plot was surveyed	Categorical	Derived from Kennedy (2010)

Spatial Predictions

Using prediction models based on NRI plots, we estimated community metric values for the entire foothill zone. We began the prediction process by using the model built with the potential stressors (Table 5) to estimate the 95% prediction interval for each vegetation type. Using these limits, we adjusted the predicted standardized metric derived from applying the best model from our factor analysis (Table 4) to each of the vegetation map units i.e. each polygon in the vegetation map. We then used the multiple regression models of environmental factors to create prediction for entire foothill zone in terms of the original community metric variables. After upper and lower limits of the standardized values were obtained, we reverted the standardization using the mean and standard deviation estimated locally for each vegetation type. Thus, there are several sources of error for each prediction: one associated with prediction using the potential stressors, another with the prediction from the factor analysis, and a third comprising the remaining error from these procedures. The width of the final confidence intervals represents the level of uncertainty in our predictions.

Temporal trends

In addition to comparing changes in spatial patterns across datasets, we explore temporal trends in plant communities in the foothill zone by using NRI plots that were re-censused between 1985 and 1999. Given that there were few re-censused plots ($n=9$) and the time and frequency of re-censuses varied, we also explored temporal changes among NRI plots grouped by the decade they were censused. While this second approach compares trends over different plots, the methods of vegetation sampling remained the same.

In order to further characterize temporal trends in plant communities after a fire, we used the Fire Effect dataset to test for overall changes in community metrics before and after a fire (for more information see Table 2). Out of the 17 fire effect plots located in the foothill zone, only 7 were suitable for this analysis, mainly because they were monitored several years before or after a fire. The number of plots available and number of census with respect to a fire event is not suitable for conducting a time series analysis. Instead we tested for differences between overall means of pre- and post-fire conditions.

Reference Conditions

Reference conditions for community composition in these vegetation types are difficult to determine because of the recent influence of invasive species and long history of disturbance since the pre-settlement period (1700-1850). Pre-settlement composition of grasses and shrubs in grasslands and hardwood forest savannas is uncertain, in part due to active management practices by Native American communities. Nevertheless, it has been suggested that the Central Valley, the central and southern Coast Ranges, and valley in southern California were dominated by native perennials like *Nassella pulchra*, *Poa secunda*, *Leymus triticoides*, or *Muhlenbergia rigens* (Clements 1920; Holland and Keil 1995; Holstein 2001; Schiffman 2007). Alternative hypotheses suggesting pre-settlement dominance by woody vegetation (Hamilton 1997; Keeley 1993, Hopkinson and Huntsinger 2005) or native annuals (Solomeschch and Barbour 2006) have also received some support. In the 19th century, non-native grasses became a dominant component of these communities due to a combination of disturbance and droughts in the first and second halves of the century (Keeley 1990, Heady 1977). Although studies agree that presence of invasive grasses has been increasing, there is less agreement on the abundance of native grasses in foothill communities.

Because some foothill communities at Sequoia National Park have been protected since 1890, they may not have experienced the same dramatic pressures as other foothill areas in the region. Grazing has been greatly reduced in the park and has not been as intense as in nearby private lands. Southern Sierra Nevada foothills have had a complicated history of grazing (Menke et al. 1996), but areas in Sequoia National Park have been free from grazing for over a century except for packhorse grazing, and some areas that experienced unauthorized cattle grazing until 1986 when fences reduced this use; trespassing grazing was fully controlled in 1997. Thus, foothill communities at the park more likely represent reference conditions for other foothill areas with respect to potential stressor levels, with the exception of fire frequencies.

Foothills at SEKI may be diverging from their natural states due to the lack of regular fire disturbance. Between 1890 and 1960, fire suppression policies were implemented in the park, but this changed with the implementation of prescribed fires in 1968 (Vankat 1977). Under fire suppression policies, oak woodlands that evolved with frequent fires might have not experienced fires for more than 70 years, while other shrubland areas might be slightly over the upper limit of their optimum fire return intervals. Recruitment of shade intolerant species, herbs, and shrubs throughout the foothills might be minimal and risk of large and severe fires might be high in some areas (Vankat 1977; Stephens 1998). In blue oak woodlands, fire suppression is thought to be responsible for the increase of non-native grasses, which was estimated to be 99% of the herb community in the foothills in the park (Parsons and Stohlgren 1989). Although there is a high level of uncertainty in the historical fire regimes of lower elevation oak woodlands, infrequent fires have contributed to a shift in community composition.

In denser oak forests and shrublands, fire suppression might have reduced the advance of non-invasive grasses, but at the expense of an increased risk of severe fire. Chaparral is resilient to fire return intervals between 25 and 100 years (Keeley 1986; Zedler 1995); typical intervals are 50 to 70 years (Minnich 1995; Zedler 1995; Conard and Weise 1998). Community composition

under optimal circumstances shifts from communities dominated by obligate seeders to communities dominated by resprouting species with a competitive advantage under light-limited conditions (Keeley et al. 2005b). Overall diversity in interior chaparral communities peaks in the years immediate after a fire, from 2.34 (SE=0.08) in the first year post-fire to 1.92 (SE=0.06) in the fourth year post-fire (Keeley et al. 2005b). In general, these communities tend to be dominated by few species with many subdominant species. In particular, these communities have a much higher proportion of herbs than shrubs, but this decreased as post-fire recovery progresses (Keeley and Fotheringham 2003).

One of the products of this chapter is to estimate the relationship between diversity metrics at the plot level and variables describing the effect of fire, grazing, and overall disturbance across the SEKI foothills. In this sense, the results of our analyses can be used as reference condition for the spatial variability of foothill vegetation diversity in areas that have been protected for more than 70 years.

Spatial and temporal patterns

Diversity throughout the Foothills

There is a strong elevation gradient in the species composition of foothill communities. Low elevation areas (<800 m a.s.l.) are dominated by oak communities (71% of the total low elevation areas), including canyon live oak-California buckeye (*Aesculus californica*) association (27%), and blue oak associations (28%). Communities dominated by chamise-shrubland alliance represent 18% at this elevation. Shrubs are present with low coverage (<5 %) and herbaceous layer is dense (approx. 80% cover) and dominated with non-native grasses, in particular, species of *Avena* and *Bromus* (mainly *B. diandrus* and *B. hordeaceus*, Keeley et al, 2003). Mid-elevation areas (between 800 and 1500 m a.s.l.) are dominated by canyon live oak (44% of the mid-elevation foothill area) and chamise shrublands (11%), which are usually located along dry slopes. Communities dominated with canyon live oak present a closed canopy ecosystem with tall conifers and oaks representing 40-100% cover and a denser shrub community. Chamise, sticky white leaf manzanita (*Arctostaphylos viscida*) and buckbush (*Ceanothus cuneatus*) are the dominant shrubs, either with uninvaded closed canopies or intermixed with an understory dominated by alien annuals (Keeley et al. 2003). At this elevation, mixed coniferous forest is mostly represented by communities dominated by white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), and canyon live oak (overall mixed coniferous coverage at mid-elevation is 6%). Mixed coniferous forests are better represented at higher elevations (1500-1830 m.a.s.l.) where 14% of the foothill area at this elevation is dominated by ponderosa pine (*Pinus ponderosa*), incense cedar, and black oak. Canyon live oak associations are most abundant at this elevation (38%), and shrublands (without a clear dominant association) are still important (13%).

When comparing across vegetation types, hardwood forests were characterized by relatively open canopies that favor growth of a dense understory layer dominated by herbaceous species. On the other hand, shrublands were usually dominated by few species that form dense, closed canopies that limit the presence of herbs herbaceous species. In particular, we found that diversity of native herbaceous species tended to be higher in hardwood forest plots than in shrubland communities (Figure 5, p-value for the comparison=0.036), which is explained in part by higher number of native species in forested areas. We also found a clear dominance of invasive grasses across all vegetation types. Finally, higher diversity of shrubs occurred in hardwood forests, while diversity of trees was lowest in these communities, reflecting the dominant role of canyon live oak and black oak.

Table 6. Summary statistics of community metrics for native herbaceous species, shrubs and trees for NRI plots across all community types. In each cell the first value indicates the mean and the two values in parentheses indicate the standard deviation and number of plots, respectively. Across the foothill zone, native herbaceous species comprised on average almost 80% of the total cover of herbaceous plants, although native grass abundance was much lower, even when considering native grass cover as a proportion of all grass cover (Figure 5). The most abundant shrub and tree were birchleaf mountain mahogany (17%) and canyon live oak (23%), respectively. Incense cedar and california black oak were also abundant tree species: they each made up an average of 7% in tree cover.

	Herbaceous plants	Shrubs	Trees
Richness	24	6.6	3.18
(# species/0.1 ha)	(13 ,77)	(3.9 ,69)	(1.658 ,61)
Shannon Index (H')	2.1	1.0	0.63
	(0.78 ,77)	(0.60 ,69)	(0.46 ,61)
Native herbaceous richness	20	-	-
(# species/0.1 ha)	(11 ,77)	-	-

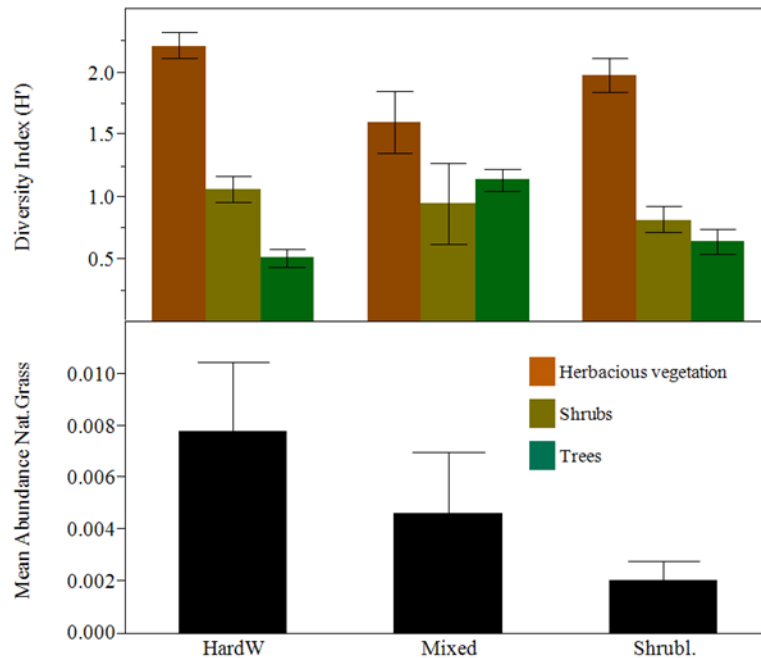


Figure 5. Average community metrics by community type using NRI plot data. Upper panel shows results for native herbaceous vegetation, shrubs, and trees, respectively; lower panel shows the mean abundance of native grasses estimated as proportion cover of native grass species/cover of all grass species. Error bars correspond to one standard error. Abbreviations: Shrbl=Shrubland, HardW=Hardwood forest, and Mixed=Mixed coniferous forest.

Spatial Patterns

Spatial variation of standardized metrics. Spatial patterns of standardized community metrics suggests that plant communities at Kings Canyon and along East Fork Kaweah are in better conditions than communities in the rest of the foothills at SEKI, although there is not a consistent pattern across metrics. Along the Middle Fork Kaweah and at higher elevations along the East Fork Kaweah watersheds, standardized values of native herbaceous Shannon diversity (H') derived from NRI plots tend to be lower than the overall average for the dataset (Figure 6).

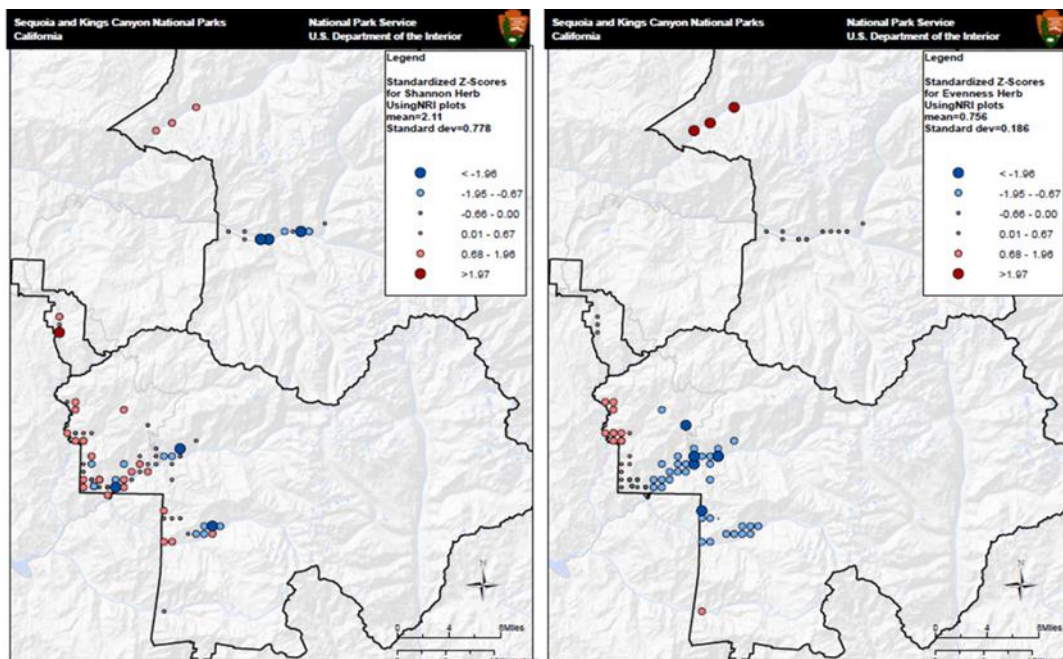


Figure 6. Standardized community metrics for native herbaceous species calculated using NRI plots. The panel on the left shows standardized Shannon diversity index (H') and the panel on the right shows Evenness (L'). Each marker represents a study plot in the corresponding dataset. Extreme colors and sizes represent increasingly significant values with respect to the overall mean for the whole dataset. Values more extreme than -1.96 or 1.96 are statistically significant at a confidence level of 95%.

Locations of highly diverse herbaceous communities do not necessarily correlate with locations of communities with high evenness, suggesting high variability in species richness. NRI plots with high dominance of a couple of herbaceous species (low evenness) were restricted to mid-elevations in the Middle Fork Kaweah, while plots that contain a more even distribution of species abundances were found at the Middle Fork Kings River unit in Kings Canyon National Park (Figure 6). Percentage of native grasses in NRI plots was low and homogenous throughout the landscape (Figure 7). Communities with high but average abundance of native grasses were found close to the administrative units of the park, along the Middle Fork Kaweah River. Lower abundances of native grasses consistently occurred along the East Fork Kaweah, although these values were not significantly different than the overall mean.

For shrubs, high diversity and evenness were located along the upper portion of both the Middle and Marble Fork Kaweah unit and at the East Fork Kaweah (Figure 8). Diversity of shrubs is low at the westernmost boundary of North Fork Kaweah watershed. High tree diversity was exclusively located along the South Fork Kings River, while plots with low diversity and high dominance were located at lower elevations along North Fork and Marble Fork Kaweah River units (Figure 9).

Importance of Temperature and Climatic Variables. Most of the detected differences across HUC-10 units are due to the preponderant role of topographic and climatic variables. Factors derived from topographic and climatic variables explain between 9 and 64% of the variability in our standardized community metrics (R^2 values in Table 7). In general, diversity and evenness of trees were strongly correlated with environmental factors (more than 60% of the variability explained), while metrics for herbs and shrubs produced more variable results.

Factor 1, which is directly proportional to elevation and precipitation and inversely proportional to temperature, predominates in our models. This factor was negatively correlated with diversity metrics for shrubs, indicating that diversity is greatest in low elevation areas with low precipitation and hot temperatures. Conversely, diversity of trees and evenness of shrubs increased with elevation and precipitation.

Native herbaceous communities showed a more complex relationship with environmental factors. Diversity of native herbaceous species tended to decrease with precipitation and increase with isolation; the trend with elevation is less clear (Table 7). Native grasses increased in evenness and proportion cover with factor 1. Given the overall diversity of herbaceous species (native and non-natives) decreased with elevation (negative correlation with factor 1; $R^2=0.18$ $P<0.0001$), while native species abundance had a weaker relationship ($R^2=0.018$ for a regression with factor 1, $P=0.02$), our results suggest that diversity here is driven by higher diversity of non-native species at lower elevations. In this sense our results agreed with previous studies in the same area that reported a negative relationship between overall understory diversity and elevation (Keeley et al. 2003).

Despite the strong explanatory power of factors associated with elevation and temperature (Factors 1 and 4), our models vary widely in terms of the percentage variability explained, which suggests that there are other important factors that are not incorporated in our analyses (Table 7). This is particularly true for native herbaceous diversity, given the low R^2 value in our best model.

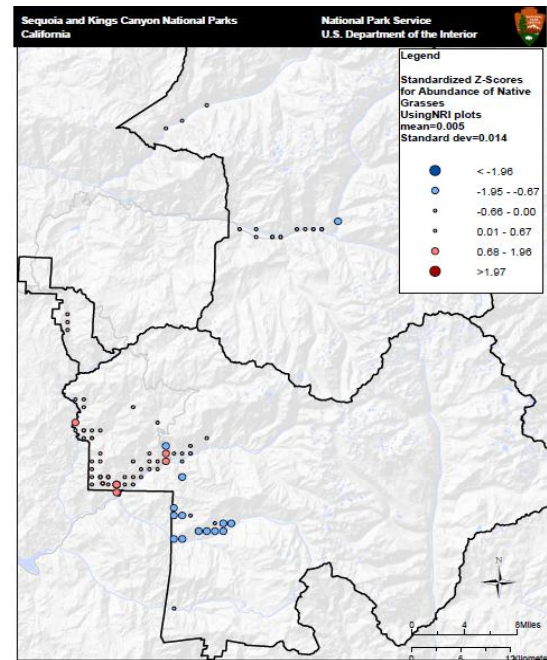


Figure 7. Standardized abundance of native grasses calculated using NRI plots. Each marker represents a study plot in the corresponding dataset. Extreme colors and sizes represent increasingly significant values with respect to the overall mean for the whole dataset.

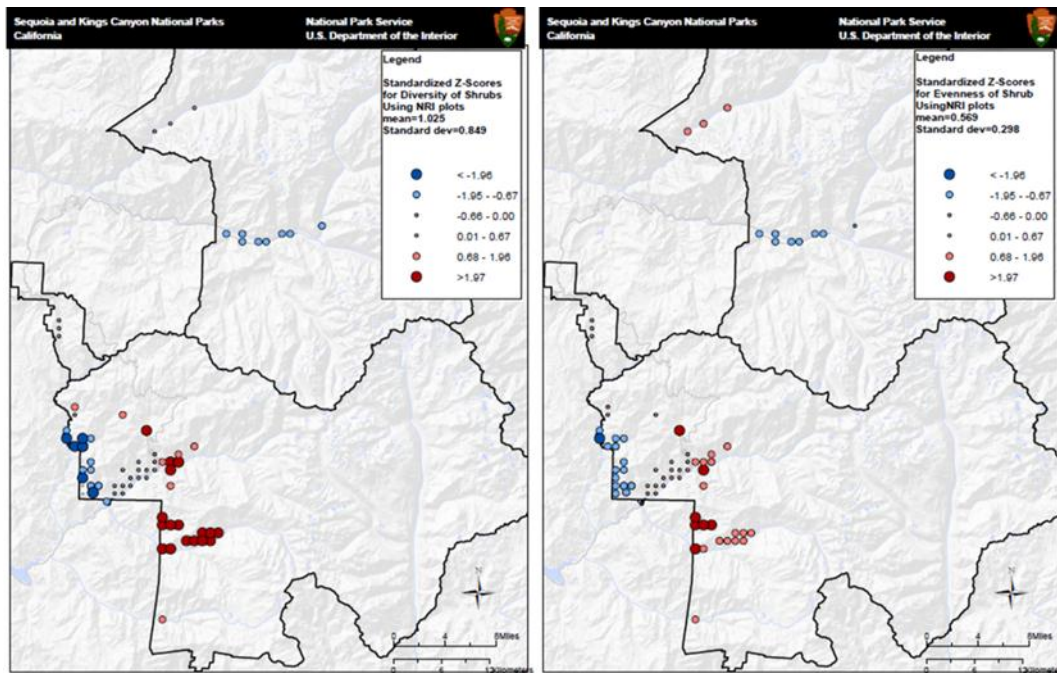


Figure 8. Standardized diversity metrics for shrubs calculated using NRI plots. The panel on the left shows standardized Shannon diversity index (H') and the panel on the right shows Evenness (L'). Each marker represents a study plot. Extreme colors and sizes represent increasingly significant values with respect to the overall mean for the whole dataset.

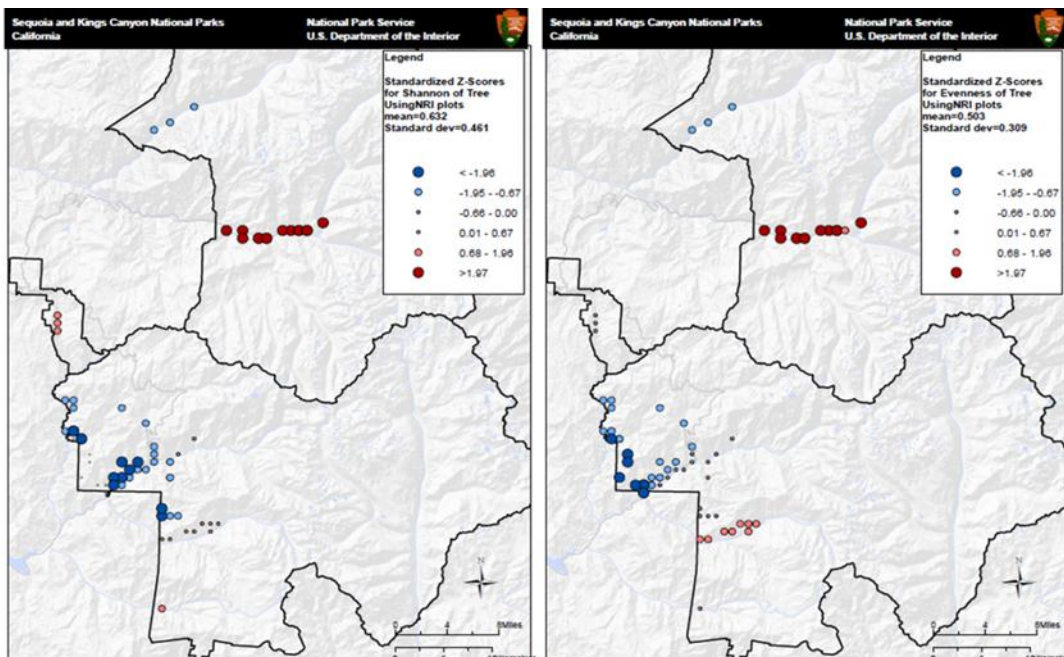


Figure 9. Standardized community metrics for trees calculated using NRI plots. The panel on the left shows standardized Shannon diversity index (H') and the panel on the right shows Evenness (L'). Each marker represents a study plot in the corresponding dataset. Extreme colors and sizes represent increasingly significant values with respect to the overall mean for the whole dataset (blue indicates areas of low diversity, red indicates areas of high diversity).

Table 7. Results from multiple regressions relating community metrics with topographic and climatic variables for NRI plots. The first half of the table summarizes results of fitting community metrics with the 5 factors derived using factor analysis on topographic and climatic variables. Model selection was performed using stepwise regression using minimum Bayesian Information Criterion (BIC). We only report the statistics associated with the best model, which include the R^2 , and the overall fit test (F-statistics). The second half of the table provides the coefficient estimates and standard errors for factors included in the best model. Correlation of each factor with the original variables is indicated by + or – symbols below the factor's name. Abbreviations: elev=elevation, prec=precipitation variables, temp=temperature, dist=distance.

	Herbs			Shrubs		Trees	
	Shannon (H')	Evennes (L')	Native Grass Relative Abundance	Shannon (H')	Evennes (L')	Shannon (H')	Evennes (L')
R^2 Adjusted	0.166	0.200	0.468	0.347	0.154	0.632	0.618
Mean of Response	-0.063	-0.367	0.067	-0.006	0.082	-0.439	-0.170
Observations (#)	93	92	93	77	77	69	69
P-value	0.054	<.0001	<.0001	<.0001	0.046	<.0001	<.0001
Estimates							
Factor1 (+elev,+prec,-temp)	-0.1407 (SE=0.0936)			-0.278 (SE=0.0348)	0.8345 (SE=0.1464)	0.5183 (SE=0.1343)	1.2624 (SE=0.1329)
Factor3 (+dist stream)				0.6571 (SE=0.2283)			
Factor4 (-elev,+temp,-dist road)	-0.3743 (SE=0.1614)	-0.7526 (SE=0.1546)	-0.2033 (SE=0.0603)				0.4727 (SE=0.1463)
Factor5 (+slope,+dist road)	0.4342 (SE=0.1239)						-0.916 (SE=0.1901)

Effect of potential stressors. After we removed variation explained by topographic and climatic variables, we explored the effects of two main potential stressors: grazing and fire. In addition, we evaluated the relationship between community metrics and attributes associated with vegetation cover loss (which we term general disturbance). In all cases, we used multiple regressions with the residuals of the previous analyses and the continuous and categorical variables describing potential stressors (Tables 3 and 5). By comparing residuals after taking into account the effect of topography and climate, we can optimize comparison between factor categories. In addition to these analyses, we also explicitly compared differences across HUC-10 units within shrublands and hardwood forests. Given that we found a low correlation between categorical variables associated with fire, grazing, or overall vegetation cover changes with environmental factors (Table 8), we are confident that our analyses or effect of potential stressors is not confounded by environmental effects.

Table 8. Correlations between environmental factors and potential stressors. Value at each cell estimates the strength in the correlation between potential stressor variables and watershed units with environmental factors. For each cell R^2 is given assuming a linear model, except for magnitude of disturbance where the correlation coefficient. See table 4 for the relationship between environmental factors and environmental variables.

	Factor 1	Factor 2	Factor 3	Factor 4	Factor 5
Watershed	0.39	0.07	0.09	0.58	0.34
Disturbance Category	0.02	0.01	0.05	0.03	0.02
Magnitude of Disturbance	0.14	-0.25	0.18	0.19	0.25
FRID	0.07	0.02	0.05	0.04	0.04
Grazing	0.35	0.05	0.00	0.00	0.03

Throughout our analysis, we found strong differences across watersheds (Table 9) regardless of potential stressor, which in several occasions explained most of the variability in the residuals after environmental characteristics were accounted for. In particular, Marble Fork showed

significantly higher abundance of native grasses given the environmental characteristics than the rest of the HUC-10 units and North Fork Kaweah showed lower than expected diversity of shrubs. Although East Fork Kaweah had lower than expected values of native grass abundance once environmental conditions were taken into account, it also had higher than expected shrub diversity for both shrublands and hardwood forests. Tree communities did not show differences across units. These results demonstrate that factors other than the ones we describe with our environmental characteristics influence spatial variation in diversity. For instance, floras may differ between HUC-10 units and our explanatory variables miss many (particularly longer-term) land use and disturbance history.

The second most important factor explaining the variability in diversity was recent disturbance as described by Landsat analysis (Kennedy 2010). We found that shrub communities where vegetation cover has been reduced by disturbance were less diverse than expected (based on environmental characteristics) compared to areas where a cover-changing disturbance was not detected (Table 9). In addition, as the magnitude of change in vegetation cover decreases (percentage of area that completely lost vegetation cover), the abundance of native grasses was reduced; this suggests the severity of disturbance detected in SEKI is high enough to favor higher abundance of invasive species. Thus, our results pointed out that overall disturbance events and not only fire have the potential to strongly influence community characteristics in SEKI.

Areas with a documented history of grazing are concentrated on the western portion the foothill zone, and are not represented in NRI sampling. To explore the potential impact of authorized and unauthorized grazing for plant communities in the park, we instead used Vegetation Mapping plots (VMA plots; Table 2) that were located in grazed areas, and compared them with VMA plots at comparable elevations that have not experienced cattle grazing. In addition, we contrast VMA plots inside and outside SEKI boundaries to explore general impact of protection policies in SEKI, which include prohibition of cattle grazing and logging. It is important to note the high uncertainty in both analyses, as a more biased dataset was used and some comparisons occurred outside NPS boundaries. In addition, we have no information on the intensity and duration of grazing history, or the intensity of wildlife grazing.

Given these limitations, one of the most interesting results related to grazing history was that native herbaceous diversity was higher than expected in previously grazed areas, while shrub communities showed lower diversity and evenness in areas with a history of grazing (Table 9). Grazed areas have experienced severe disturbance in the past years based on vegetation cover change, and moderate to low fire return interval departures (FRID is usually 3 or 4 in these areas, although these estimates are highly uncertain), circumstances that confound any potential effect of grazing. Despite this, it is clear from our results that native herbaceous communities in areas that have experienced cattle grazing until 1997 are no worse than in other areas throughout the foothills. Keeley found similar results for the SEKI foothills, which suggests that impact of grazing have signatures lasting several years (Keeley et al. 2003) or the level of grazing experienced was within levels that favored native species (Marty 2005; Hayes and Holl 2003; DiTomaso et al. 2008, Jackson and Allen-Diaz 2006; Huntsinger et al. 2007).

Diversity in plant communities in areas outside the park did not differ from comparable areas inside the park where grazing was restricted (Figure 10). In fact, we detected a significantly higher abundance of native grasses and Shannon diversity values for native herbaceous species outside the park than in comparable areas inside the park ($\text{Abundance}_{\text{out}}=0.04$ SE=0.018, $\text{Abundance}_{\text{in}}=0.001$ SE=0.02 with p-value <0.05 for the t-statistics; $H'_{\text{out}}=0.54$ SE=0.055, $H'_{\text{in}}=0.25$, SE=0.61). Trees and shrub diversity were similar in both locations. However, these differences could also be explained by differences in environmental conditions between areas (Figure 10). Furthermore, it is important to note that East Fork Kaweah plots ranked the last for abundance of native grasses in comparisons across watersheds (Table 9). More research specific to grazing impacts is needed to determine the influence of grazing history on the conditions of foothills areas in SEKI.

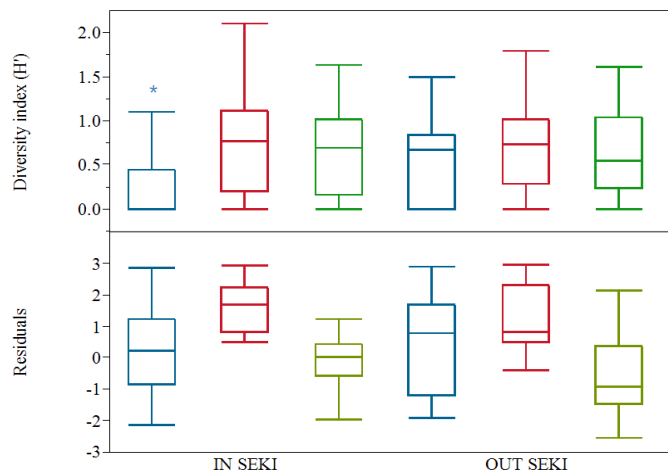


Figure 10. Comparison between areas inside SEKI and areas outside the park. Upper panel shows the Shannon diversity index (H') for herbaceous vegetation (blue), shrubs (red), and trees (green). Lower panel shows the results after any correlations with environmental characteristics were taken into account. Within each vegetation group, we compared areas in and outside the park using t-tests; an asterisk indicates significant differences. For this comparison we used Vegetation mapping plots adjacent to the park on the East Fork Kaweah unit (Figure 1).

Due to limited understanding of historical fire regimes in the foothills, analyses relating FRID scores are uncertain, particularly in lowest elevation foothill areas. Despite the expectation that severe and extreme fire return interval departures will impact plant community diversity, particularly in shrublands, we found a weak signal of this factor in our analysis (Table 8). More research is needed to better understand historical fire regimes and the effects of departures from these regimes.

In chaparral, fires are typically high severity crown fires that could reset communities to bareground conditions. Under increased presence of invasive grasses, one of the major risks in these communities is strong departure from historical fire community fluctuations promoted by frequent fires (Keeley et al. 2008). While fire severity and frequency control which species persist in the community, pre-fire community composition will control the actual composition of communities after a fire (Keeley et al. 2003). In addition, community composition after fire changes as canopy closure limits regeneration of light intolerant species, and seed obligate species are replaced by resprouters (Keeley and Forthingham, 2003). Given that hardwood forests communities are dominated by fire tolerant species like blue oak, changes in plant diversity after disturbance are mostly driven by year to year changes in herbaceous species. The upper elevation hardwood forests dominated by canyon live oak and black oak are considered relatively stable (Bolsinger 1988).

Analysis of community response to fire using Fire Effect Plots (Table 2) did not show any signs of increased abundance of invasive species. Diversity of native herbaceous species was significantly higher after a fire ($\text{Mean}_{\text{pre}}=0.79$ $\text{SE}=0.19$, $\text{Mean}_{\text{post}}=2.05$ $\text{SE}=0.29$, $p\text{-value}_{\text{t-pair-test}}=0.002$), most likely indicating the positive effect of canopy openings due to fire (Keeley et al. 2003). Diversity of shrubs significantly decreased after fire (for diversity $\text{Mean}_{\text{pre}}=0.88$ $\text{SE}=0.19$; $\text{Mean}_{\text{post}}=0.18$ $\text{SE}=0.17$), while tree diversity did not change (for diversity $\text{Mean}_{\text{pre}}=0.63$ $\text{SE}=0.09$; $\text{Mean}_{\text{post}}=0.44$ $\text{SE}=0.11$). In shrublands, the high proportion of herbs that are observed after fire yields to increased dominance of shrubs as recovery progresses (Keeley and Fotheringham 2003b). Although shrubland communities often recover within 2-3 years of a fire (Keeley et al. 2005a; Lozano et al. 2010), spatial variability can be high (Lozano et al. 2010) depending on pre-fire community composition (Keeley et al. 2005a). This limits pattern detection when only few sites are available, the case in our analysis.

Table 9. Results from multiple comparisons of residuals of diversity metrics (once environmental correlations have been accounted for) with potential stressors (these variables are explained in Table 5). We also explored differences among watershed units. We indicate the significant differences, and the percentage of residual variability explained is included in parentheses. For watersheds, multiple comparisons with Bonferroni corrections were conducted. Given that we did not find any significant effects of disturbance duration and fire (FRI Departure), we have not included them in this table. Because sample size was too low to conduct grazing history comparisons based on NRI plots, results for grazing (last column) are based on analyses with VMA plots. Acronyms for watershed units are the same as in Table 1

Vegetation type	Layer	Variable	Watershed	Disturbance		Grazing*
				Category	Magnitude	Category
Shrubland	Herbs	Shannon	n.s	n.s.	n.s.	n.s.
		Evenness	rest<NFKw (56%)	n.s.	n.s.	n.s.
		Abundance Native grasses	EFKw<rest<MbFKw (22%)	n.s.	Decrease with disturbance magnitude (26%)	n.s.
	Shrubs	Shannon	NFKw<rest<EFKw (45%)	NoDist>Dis t(1%)	Increase with disturbance Magnitude (1%)	n.s.
		Evenness	NFKw<rest<EFKw (43%)	NoDist>Dist (5.6%)	Increase with disturbance Magnitude (1%)	n.s.
		Shannon	n.s	n.s.	n.s.	NoGraz<Graz (4.1%)
Hardwood forest	Herbs	Evenness	rest<NFKw (72%)	n.s.	n.s.	NoGraz<Graz in NoDist (Graz=9.4%)
		Abundance Native grasses	n.s	n.s.	n.s.	NoGraz<Graz (5.1%)
		Shannon	MFKw<rest (70%)	n.s.	n.s.	Graz<NoGraz (1.1%)
	Shrubs	Evenness	NFKw<rest(74%)	n.s.	n.s.	n.s.
		Shannon	n.s	n.s.	n.s.	n.s.
		Evenness	n.s	n.s.	n.s.	n.s.
	Trees	Shannon	n.s	n.s.	n.s.	n.s.

Spatial Predictions

At the extent of the foothill zone, our analyses predict that differences in diversity of native herbaceous plants are mostly associated with vegetation types. Thus, areas with more diverse native herbaceous communities are found where hardwood forests predominate (Figure 11); lowest diversity is predicted for Middle and East Kaweah units. Abundance of native grasses is highest in the Middle Fork Kings River and in the west and south borders of North and Marble Kaweah river units. Lowest abundance of invasive grasses is predicted at higher elevations (Figure 12).

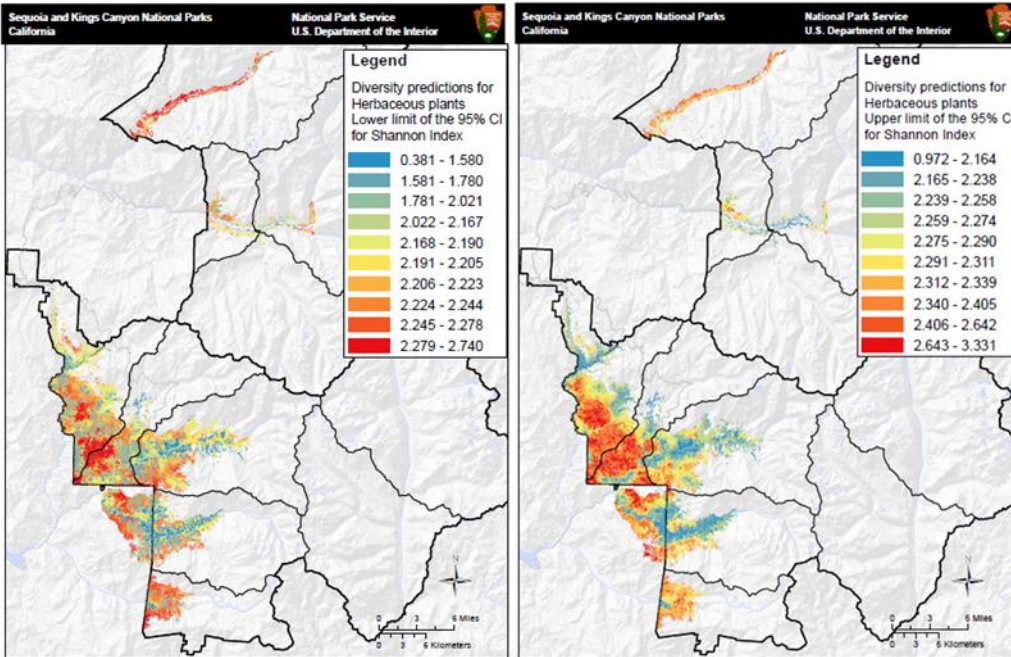


Figure 11. Predicted diversity (H') values for native herbs using NRI plots. Values were predicted for each vegetation cover separately using the best models after testing for effects of three stressors (disturbance, fire, and grazing) and topographic and climatic variables. Left and right panels show lower and upper limits for a 95% confidence interval for the mean.

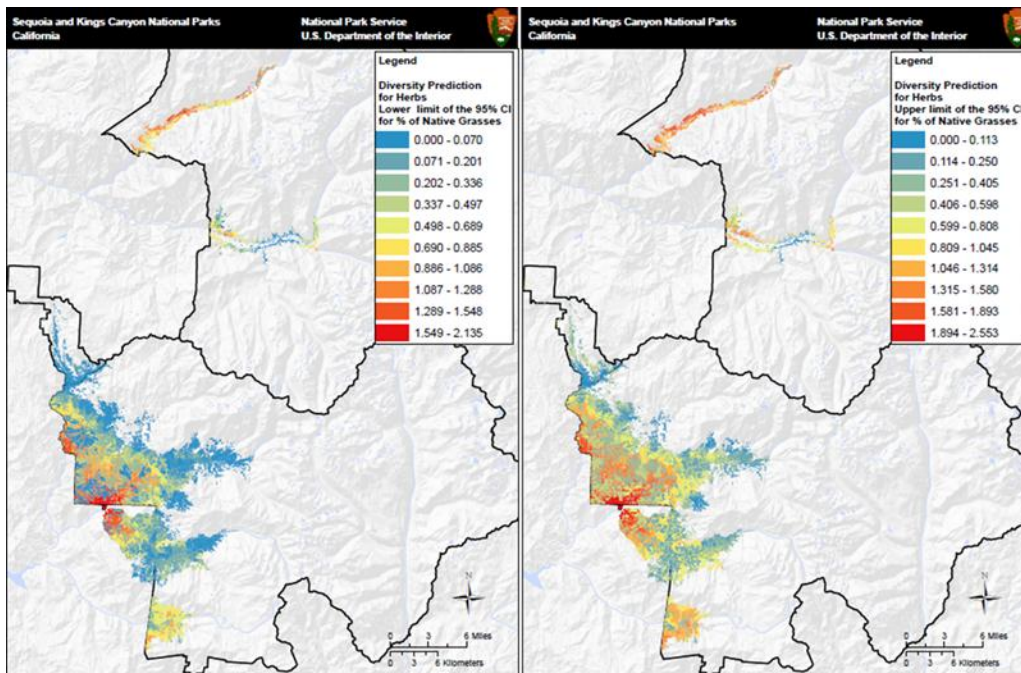


Figure 12. Predicted abundance of native grasses using NRI plots. Values were predicted for each vegetation type separately using the best models after testing for effects three stressors (disturbance, fire, and grazing) and topographic and climatic variables. Left and right panels show lower and upper limits for a 95% confidence interval for the mean.

Diverse communities of shrubs are located mainly at higher elevations along the Middle Fork Kings River and the East Fork Kaweah units; in addition, some areas throughout the Sequoia National park have high predictions of shrub diversity (Figure 13). On the other hand, tree communities are more diverse in Kings Canyon National Park and at higher elevations in the transitions between foothill communities and coniferous forests (Figure 14). Overall, we detected that areas with plant communities in the poorest condition relative to the rest of the SEKI foothills are located along Marble and Middle Fork Kaweah.

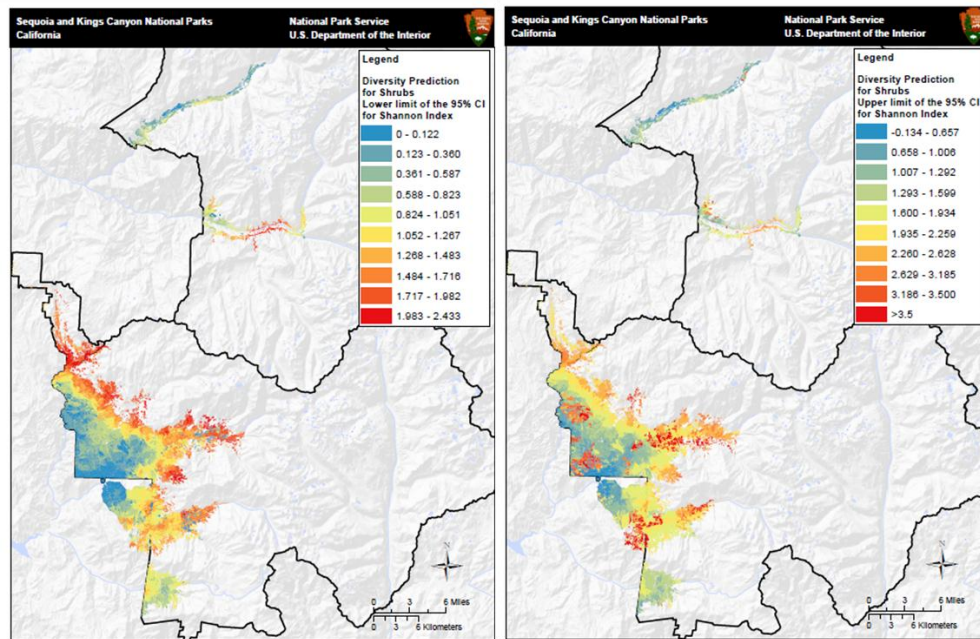


Figure 13. Predicted shrub diversity (H') using NRI plots. Values were predicted for each vegetation cover separately using the best models after testing for effects of three stressors (disturbance, fire, and grazing) and topographic and climatic variables. Left and right panels show lower and upper limits for a 95% confidence interval for the mean.

The value of these predictions for the entire foothill zone resides in the spatial patterns they suggest more than in the absolute value they represent. Similarly, given that standard error associated with our prediction increases as diversity values increase for most of our metrics, we have more confidence for low native herb diversity values than for high values; this is particularly the case for Kings Canyon National Park where our confidence is relatively low.

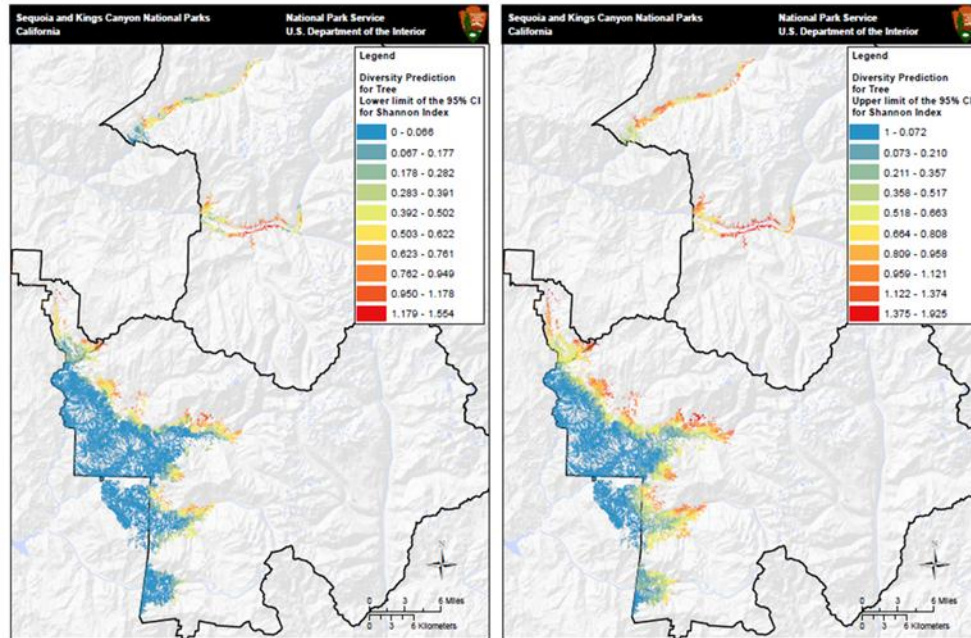


Figure 14. Predicted tree diversity (H') using NRI plots. Values were predicted for each vegetation cover separately using the best models after testing for effects of three stressors (disturbance, fire, and grazing) and topographic and climatic variables. Left and right panels show lower and upper limits for a 95% confidence interval for the mean.

Temporal Trends

Analysis of NRI plots censused more than once between the 1980s and the 1990s indicates a general decline in plant community diversity, although this decline was only significant for herbaceous species diversity. For herbaceous species, diversity decreased at a rate of -0.02676 per year ($SE=0.0084$, $R^2=0.9171$), while communities became less even at a rate of -0.0130 per year ($SE=0.003$, $R^2=0.6914$). We did not detect any differences associated with HUC-10 units, although we could only test for differences within Sequoia National Park. In addition, these trends were similar across vegetation cover types (shrub and hardwood forest). Within plot variability was very high for both metrics; for diversity we found that plot variability constituted 88% of the total variability, and for evenness, plot variability constituted 50% of the total variability. Thus, although there was a significant relationship of declining native herb diversity with time, high intra-plot variability limits our ability to extrapolate.

When plot metrics were analyzed by decade, we did not find any evidence of temporal trends, except for a decline in diversity of shrubs, (Figure 15, $H'_{1980's}=1.02$ $SE=0.08$, $H'_{1990's}=0.67$ $SE=0.14$, t -test p -value=0.038).

Uncertainty Analysis

Several characteristics of the datasets used limit the power of our analysis to accurately describe plant community conditions throughout the foothills. Sample size is the major limitation of the NRI dataset. Given that we used it in all our spatial analyses and for developing our predictions, there is a high level of uncertainty in HUC-10 units where no NRI were located. In particular, areas along the Roaring river, South Fork Kaweah, North Fork Tule, Mill Flat Creek, and Dye Creek units did not contain NRI plots.

We limit our analyses to community level metrics as we have little confidence in considering species-specific response to potential stressors. We also are more uncertain about comparisons between areas that contain different communities compared to comparisons among areas with similar flora, e.g. King Canyon vs. Sequoia National park.

Native grasses have received relatively little monitoring attention and, while the datasets we use in our analysis do describe native grass abundance, sampling should be more intensive given the low abundance of this vegetation group. More specific mapping and distributional studies aimed at understanding native grass component of the foothills vegetation is needed.

The datasets available also had little resolution to determine the regeneration status of oak woodlands, as seedlings are not measured in most of the vegetation surveys. Thus, we have very limited ability to speculate about the status of oak woodlands in our analysis. Accordingly, in the next section of this chapter, we present a literature review focused on this community type.

In addition to this level of uncertainty, our analyses also have limited power to detect effects of grazing and fire return interval departures – the two major potential stressors of the foothills. We have most confidence in the general disturbance patterns associated with cover loss recorded via Landsat analyses (Kennedy 2010). Areas that had been grazed in the past showed high disturbance levels and moderate FRI departure (Figures 2 and 3). Thus, high spatial correlation between potential stressors limits our ability to separate their effect. In addition, we have very course-level information on grazing history available for analysis. Finally, we have limited understanding of historical fire regimes in the lower foothills and FRID values for these areas are

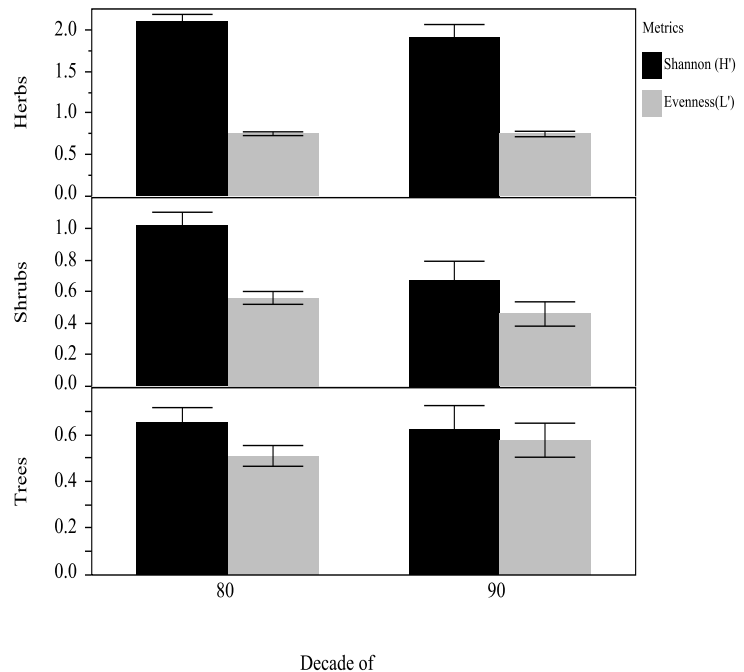


Figure 15. Changes in time of community metrics for NRI plots. Plots were grouped by the censused year in decadal intervals, and comparisons were performed on the residuals after our spatial analysis.

highly uncertain. More research is needed to understand the impacts of both of these potential stressors.

Regeneration of Blue Oak Woodlands

The objective of this section is to explore the current state of knowledge on factors impacting the integrity and vulnerability of the foothill vegetation within SEKI. The review concentrates on research that focuses on the past and present status of blue oak woodland and factors that affect regeneration of blue oaks, as this is one of the most dominant vegetation communities and important components of the foothill region.

It has been suggested that fire, grazing, and invasive species are particularly important human instigated factors affecting the regeneration of blue oaks. Historically, oaks have been removed for commercial fuel wood harvesting, to improve forage for grazing, to expand rangeland, to increase agriculture, and to create space for urban development (Bolsinger 1988; Alagona 2008). Although blue oaks survive well after fire (Haggerty 1994) and fires cause sprout development (Mensing 1992), fire history has been correlated with changes in population structure in oak savannas (McClaran 1986; Mensing 1992). In addition, grazing pressure has been negatively correlated with blue oak regeneration (Hall et al. 1992; Standiford et al. 1997; Swiecki et al. 1997a) and the invasion of exotic species (Gordon et al. 1989; Gordon and Rice 1993; McCreary and Tecklin 1997). Although there has been a massive amount of work in regeneration of blue oak and several recent reviews have evaluated the blue oak regeneration problem for California in general (see Swiecki and Bernhardt 1998, Allen-Diaz and Jackson 2005, Tyler et al. 2006), we specifically explored the potential threats to blue oak populations in SEKI. We have included in this review only work that have been conducted in areas comparable to SEKI, i.e. other foothill areas along the central and southern Sierras. Nevertheless, to complement our assessment of blue oak critical transitions and threats, we have summarized major findings in other areas in California where blue oak communities have been studied. For this reason although our literature review is comprehensive for areas comparable to SEKI, it does not reference directly studies conducted in other areas. Naturally occurring controls to blue oak regeneration will also be evaluated to give reasons for the current regeneration issue.

Regeneration processes

Regeneration of individuals or stands of trees refers to the renewal of adults lost to mortality via recruitment of new individuals, or the regrowth of individuals damaged by fire or other disturbance (Bartolome et al. 1987). In 1908, Sudworth noted that regeneration rates in several oak species appeared to be insufficient to replace natural mortality and removal of adult trees (Sudworth 1908). This initial observation generated demographic studies of blue oak populations over the past several decades, in an attempt to fully comprehend the regeneration problem.

During a comprehensive evaluation of blue oak cover changes throughout California, Davis et al. (1995) found that while some sites had noticeable decreases in tree cover, this loss was compensated by expanding tree cover in other regions. While large changes were recorded in individual regions sampled, the mean overall tree cover remained virtually unchanged from 1940 to 1998. They measured changes in extant blue oak foothill woodland cover and densities in 708 sites throughout the state using aerial photographs from the 1940s and the late 1980s.

Although their results indicate stable tree cover throughout the state for the years monitored, this does not necessarily suggest that oak woodland cover is under similar conditions today. In addition, there is a strong spatial variability in blue oak regeneration and different areas might be experiencing the effect of different potential stressors.

Locally, grazing, fire, and competition from invasive annual grasses have been suggested as determinants of blue oak seedling and sapling growth and survival, but there is little information about the relative importance of these factors (Phillips et al. 1996; Phillips et al. 2007), in particular in the long-term. As mentioned by McClaran (1985), the transition from seedling to sapling to tree appears to be more crucial in limiting regeneration than acorn germination and early seedling survival alone. Standiford et al. 1997 found that although seedlings can be found in blue oak woodlands across the state, saplings between 1.5 to 3 m tall are generally absent. Therefore, we focused on disturbances that restrain seedling regeneration into the sapling stage of life. In the next section we review the state of knowledge of regeneration events in areas pertinent to SEKI

In SEKI, Vankat and Major (1978) reported a significant increase in blue oak cover since the late 1980s that was interpreted as the result of peak regeneration 1860 and 1880, corresponding to a period in time European settlements were expanding in the region (Vankat and Major 1978). Based on their results, they hypothesized that successful regeneration and subsequent increases in blue oak densities were the result of livestock grazing introduced by European settlers in the late 1800s. Intensive grazing removed herbaceous understory vegetation, which reduced fuel loads and thus fire frequency.

Two other studies in blue oak woodlands agree with the time at which regeneration has declined, although they found no positive effect of grazing. McClaran (1986) found a continuous representation of blue oak trees that had been established between 1860 and 1930 in Tulare and Yuba Counties. He divided his study between two sites, one which had been grazed, and another which had not. He found that regeneration in the ungrazed site peaked between 1870 and 1900, and noted a distinct lack of regeneration following the 1920s. On the other hand, in the grazed sites regeneration peaked between 1910 and 1930, with few new trees appearing after the 1940s.

Although McClaran found no clear relationship between grazing and regeneration, he did discover that fire had a positive relationship with increased oak densities. From 1850 to 1940, 70-85% of the blue oak trees that had established did so within one year of a fire. In the Tehachapi Mountains of Kern County, Mensing (1988) found a continuous establishment of trees from 1570 to 1856, a regeneration peak in 1856, and a noticeable lack of regeneration following the 1860s. Specifically, he found evidence of an increase in fire frequency from 1853 to 1856 and a distinct decrease in fires after 1863. He concluded that dry conditions in the mid-1850s stimulated an increase in fire frequency which resulted in a high percentage of multiple-stemmed and -centered trees, suggesting that the burst of regeneration in 1856 was primarily the result of sprouting encouraged by fire. Mensing pointed out that the lack of regeneration following the 1860s coincides with the introduction of livestock grazing on the site in 1864, and more likely reflect summer grazing on oak seedling. Thus although fire has been congruently identified as factor promoting demographic changes in oak populations, grazing effects are less clear.

Increased regeneration after fire is a direct result of blue oak capacity to resprout following disturbance such as fire, grazing, or cutting. Fire mechanically removes the tops of small blue oak trees and stimulates post fire sprouting (McClaran and Bartolome 1989, Mensing 1988, Arevalo et al. 2009). However, the success of sprout sapling establishment can vary widely between locations, and between different years at a location (Swiecki and Berndhart 2001). In addition there is a size limit for individuals to resprout, which reduces resilience of population to fire as individuals get older. Haggerty (1994) followed an intense, crown scorching arson fire in SEKI and made subsequent visits in 1988 and 1989 to record new mortality, sprouting, and formation of new scars on blue oak trees. Adult blue oaks were able to withstand this intense fire. The survival rate was 93% (without considering seedling and saplings). However, successful sprouting did not occur in blue oak individuals that were greater than 20 cm diameter at breast height (DBH). Johnson (1977) also found that degree of sprouting decreases with increasing stump diameter. McCreary et al (2006) found that blue oak sprouts generally grow more rapidly than seedling growing from acorns. He found that after two years of disturbance by fire, an average of 90 cm in growth of stump sprouts were well above the browse zone (1.5m). In this sense, fire is a factor controlling changes in oak population structure, although these changes do not imply renewal of oak population by acorn regeneration.

In a comprehensive study of blue oak populations at SEKI, Swiecki et al. (1993, 1997a and b) calculated an adult mortality rate of 25% over a 42 year period. They calculated net regeneration by comparing blue oak sapling regeneration with adult mortality rate, and found a small portion of plots might have a net gain (less than 8%), while most plots will not change (47%) or might have a net loss (45%). Swiecki et al. (1993, 1997a, 1997b) also established that saplings are more likely to occur in the open areas than under canopy (50% of these saplings were found in the open, 27% at the canopy edge, and 22% below the canopy). These results suggest that in Sequoia National Park, saplings are much more likely to occur in the open than under tree canopies, in contrast to small seedlings, which are most likely to be found beneath the canopy of the parent tree (Muick and Bartolome 1987). Swiecki et al. (1993) hypothesize that this shift in species occurrence within characteristic sites results when seedling regeneration beneath the canopy is released through canopy mortality or removal, creating environmental conditions favorable for saplings.

Additional studies in SEKI have suggested that grazing and climate changes more than fire could be responsible of lack of regeneration. In order to understand the effects of prescribed burning on woodland and forest composition and structure, 76 plots were established in the park in 1969 and re-censused in 1998 (Roy and Vankat, 1999). They compared population structure across areas with different fire history and found that in all areas tree densities had decreased in blue oak woodland by 19% (111 trees/ha) although it remained the dominant tree with 12% of all trees censused (78 trees/ha). Size distribution of blue oaks also revealed changes from 1969 to 1996 with a statistically significant decrease in the proportion of trees in the 10 to 19 cm DBH height size class, where mortality was 15%. At that time they did not detect regeneration of new individuals in any of the areas censused. They concluded that blue oak woodlands were experiencing a reversal from previous regeneration trends (peak between 1860 and 1890) and that fire was not a determinant factor in this reversal in the park; they further suggested that

grazing reduction and climate changes could have been responsible. Nevertheless, this conclusion does not consider a formal test of different grazing levels in the park.

In SEKI, saplings appear to be absent from areas of heavy grazing pressure of livestock browsing; in fact in these areas they identified chronic vegetation browsing as the main factor limiting sapling regeneration and they did not find a significant effect of recent fire events on sapling regeneration (Swiecki et al. 1993 and 1997b), although they tested several topographic and environmental variables. They studied the distribution of blue oak saplings at the stand level in 100 plots distributed along the Marble and Middle Forks of the Kaweah River. Of these plots, 83 were within the main pasture use for authorized grazing (grazed year round since 1977 at approximately 0.9 animal-month/acre/year), 15 were within the holding area (grazed between May and October at 5 animal-month/acre/year), and 2 were outside the pasture (light occasional grazing since 1962). This study reported that 22% of the plots had low sapling regeneration, 26% had moderate regeneration, and 4% had high regeneration (Swiecki et al. 1993 and 1997b). Thus, although sapling regeneration occurs in these areas, grazing indirectly increased the chance of sapling mortality, for smaller sapling classes are susceptible to being severely damaged or killed by wild predators, fires, or other agents that would not seriously affect larger saplings or trees.

In addition, time of grazing and the influence of wildlife grazing can be significant factors. Hall et al (1992) found a strong effect of time of grazing. Winter grazing at low and medium stock densities was the least damaging of their treatments, while spring or summer grazing at high stock densities was most damaging. In addition, other studies have suggested deer browsing as is an important factor in the lack of regeneration in oak populations as deer populations have increased with a consequent intensification of browsing pressures (White 1966, Griffin 1973, Swiecki et al. 1993, Kuhn and Johnson 2008). Damage to oak seedlings under oak trees may be greater when cattle or deer are present because they are often attracted to higher forage levels, forage quality and acorns present beneath the canopy.

Despite this uncertainty concerning natural levels of grazing variability, regeneration in areas not experiencing intense grazing or under active fire management might be more strongly controlled by variability in environmental conditions in space and time. Allen-Diaz and Bartolome (1992) examined the effects of prescribed fire and sheep grazing on naturally regenerating blue oak seedlings. They found that recent low-intensity prescribed burns applied during fall months following rain prevented grass and litter at the soil surface from being completely consumed by fire, and that sheep grazing did not affect regeneration, survival, or growth of blue oak seedlings over a four year period.

Spatial variability in oak regeneration, probably driven by local environmental conditions, seems to be the norm at regional and local scales. More recent studies in SEKI have not found the same sapling regeneration potential reported in Swiecki et al (1997b). Results from a CALFED report by Stahle et al. (2010) assessed seedling to adult ratios as an alternative to evaluating regeneration potential within the Kaweah River drainage of Sequoia National Park. They sampled a total of 103 blue oak trees per hectare and found no blue oak seedlings within their study site.

Our analysis of blue oak woodlands in the foothills also indicates strong spatial variability in population characteristics. We found that trees are relatively large in diameter and regeneration is occurring in some areas (Table 10). Our estimated average tree diameter is around 22.8 cm, and is consistent with reports across datasets. Nevertheless, this mean diameter is significantly lower than tree stature in old growth forests. Stahle et al. (2013) visited two old growth forests located close to SEKI and found that average tree diameter was about 39 cm. These differences reflect the presence of very large trees that recruited before 1600s in plots sampled by Stahl, rather than a difference in regeneration chronologies. In fact, peak regeneration for the park is between 1860 and 1890 (Vankat and Major, 1978) which coincides with regeneration episodes in plots outside the park (peak regeneration between 1860 and 1900, Stahle et al. 2013), and in general with peak regeneration across the state (McClaran 1986; Mensing, 1992; Stahle, 2010 and 2013).

The sapling to adult ratio can be used as a proxy for evaluating regeneration. Using this measure, we found that the ratio in some areas of SEKI was higher than one, indicating that successful regeneration of saplings could be occurring in some areas of the park. In particular, two locations along Middle Fork Kaweah (3-3.5 saplings/adult) and in one location at the west border of the park, close to Yucca Mountain (3 saplings/adult), show high ratios. These values contrast with values reported by Stahle in old growth forests (0-0.02 sapling/adult) and 0-0.02 sapling/adult in the other 15 NRI plots evaluated. Thus, although overall regeneration is low, there are areas inside the park with active sapling regeneration. Considering the strong signature of environmental variables in community composition; we would recommend to conduct a spatial evaluation of blue oak populations at SEKI.

Table 10. Summary of blue oak metrics at SEKI. Results are grouped by dataset where NRI = Nature Resource Inventory plots, VMA = Vegetation mapping areas, and VM Full plots = the second set of the Vegetation mapping plots. In each cell the first value indicates the mean and the two values in parentheses indicate the standard deviation and number of plots, respectively. DBH = diameter at breast height (in cm).

	Fire Effects	NRI	Stahle	Vankat-Roy (1966)	Vankat-Roy (1996)	VM Full Plots
Mean DBH	28 (4.6 ,2)	23 (5.1 ,18)	40 (2.8 ,2)	22 (7.3,14)	26 (7.0, 14)	24 (4.9 ,12)
Min DBH	14 (8.6 ,2)	13 (9.2 ,18)	15 (4.1 ,2)	13 (4.3,14)	16 (4.9,14)	15 (4.1 ,12)
Max DBH	38 (4.7 ,2)	39 (13 ,18)	78 (2.0 ,2)	34 (17,14)	39 (16,14)	41 (15 ,12)
Sapling/ adult ratio	NA	0.54 (1.7 ,18)	NA	NA	NA	NA

Studies in areas close to SEKI have found reduced regeneration in blue oak population. Phillips et al. (1997) conducted a survey in Tulare and Kern Counties in California to evaluate blue oak regeneration factors including tree size class, acorn quality, and natural seedling survival. They found many individuals in the seedling and mature tree size classes, but consistently fewer in the sapling and pole size classes. Similarly, a technical report from the Pacific Northwest Research Station (Barrett and Waddell, 2006) explored the diameter distribution of blue oaks from three successive forest inventories in California (1981-1984, 1991-1994, 2001-2005). They concluded that diameter distributions have departed from the usual inverted J-shaped, which is characteristic of trees that are successfully regenerating. Their report concludes that sparse regeneration and low densities of small saplings for blue oak have been consistently observed for three decades of forest inventory.

Studies all agree in identifying a current lack of regeneration across the state, and in particular for populations at SEKI. Seedlings and saplings are rare in blue oak populations, but regeneration is spatially variable (Harvey 1989), as it is also observed in SEKI. Tree cover increased at SEKI since the 1920s, although density decreased up to 1996 (Vankat and Major 1978, Roy and Vankat 1999). Statewide, density has remained unchanged in most areas (Holzman 1993; Swiecki et al 1993), and changes in cover are variable (Davis et al. 1995). In general, fire seems to have been an important factor explaining pulse of regeneration and increased densities in many locations throughout the state. Grazing has been identified as a factor reducing seedling and sapling growth and survival, but only under intense levels.

Although these studies provide a good indication of changes in past regeneration and the present status of blue oak distribution, they do not differentiate between factors affecting transitions at each life stage of the blue oak. In order to understand how blue oak community composition may change in the future under different management practices and with a changing climate, it is necessary to consider factors that constrain oak regeneration. Specifically, we recommend further demographic modeling to investigate the factors that influence survival and mortality at each stage of the blue oak life cycle: acorn, seedling, sapling, sprout, and mature, seed bearing tree.

Other Factors affecting natural blue oak regeneration

Oak species produce seed crops that vary widely in quantity from one year to the next. Episodes of high acorn production occur through a process known as mast-fruiting, and are usually separated by several years of low acorn production. A study of acorn production patterns in five species of oak by Koenig et al (1994) found that valley oak (*Quercus lobata*) and blue oak had very similar acorn production patterns across years and that large acorn production one year was often followed by low acorn production the next. In fact, blue oaks exhibit a strong synchrony over large geographic scales of up to 135 km (Koenig et al 1999). Their results demonstrate that a local mass-fruiting event can influence acorn production over a large spatial scale, and can increase the probability of germination. Unfortunately, regular masting cycles and the processes that influence masting have not yet been identified for any species in the genus *Quercus*.

Several factors affect blue oak probability of germination and survival in the first years, most of these applicable to SEKI populations. Acorns tend to land beneath the canopy of their parent tree, but can be planted beyond the canopy through secondary dispersal by seed caching animals. Blue oak acorns are viable for one year and within that time will germinate when moisture is available. A survey conducted in Kern County, assessed environmental factors that contribute to acorn quality and survival (Phillips 1992). The study exposed water drainage as an important environmental parameter that influences acorn weight. Elevation also appeared to greatly influence blue oak acorn germination. Acorns at lower elevations (620 meters) had the lowest germination rates of 19%. The highest germination rates of 82% occurred at mid-elevations, and this rate decreased to 46.2% with slight elevation gain up to 1,400 meters (Phillips 1992). Borchert et al. (1989) also found an altitudinal gradient related with moist conditions, with mesic sites at lower elevations having greater levels of seedling regeneration than cooler, higher elevation sites. Thus these results emphasize the importance of soil moisture in successful

germination. In fact, Borchert et al. (1989) found that regeneration rates for buried acorns were twice those for surface-sown acorns due to improved germination and reduced predation. Tecklin and McCreary (1991) found that acorn size significantly influenced survival, germination, and growth. Large acorns produced taller seedlings with heavier total root weight. These results suggest that increased growth and survival may be obtained by selecting large acorns for reproduction.

Predation by a variety of small and large mammals, birds, insects, and fungi also poses a natural threat to acorn survival. Foraging by livestock has proven to be a threat to acorn survival, since cattle often browse on fallen acorns. Within SEKI, the impact of livestock grazing on acorn survival is not an issue since grazing is no longer allowed in the park, and predation by wildlife is considered a greater threat to acorns. Past research on acorn predation has shown that many mammals, the most commonly mentioned being pocket gophers, mice, and deer, pose a threat to acorn survival. Despite this, there is no evidence that consumption of acorns is a primary cause for poor regeneration (Schmidt and Timm 1991).

Environmental conditions and microclimate are two other factors influencing natural regeneration in blue oak populations. Although acorns tend to accumulate below adult canopies, germination is not higher in these areas in general; it could be higher under drier conditions. Borchert et al. (1989) determined that oak canopy cover and herbaceous understory seemed to be significantly related to seedling regeneration, but these effects were minor and varied between sites. The oak understory provides cool, moist microhabitats compared to open grassland microsites (Parker and Muller 1982; Borchert et al. 1989). This suggests that understory sites may provide ideal microclimates for seedling regeneration only in xeric locations, while understory sites may be less beneficial in more mesic locations. Similarly, Standiford et al. (1991) found that although seedlings are usually found in blue oak populations, fewer seedlings and sapling have been found at low rainfall zones (230-380 mm/year) in oak woodland ranges within the foothills (lower than 1000 m.a.s.l.). Thus, moisture and geographic features appear to affect the emergence of blue oak seedlings. Although blue oaks are the most drought-tolerant of all the oaks, they still appear to be sensitive to periods of drought.

Mortality rates of blue oak seedlings and saplings are important in understanding future rates of regeneration. A 2007 survey by Phillips et al. evaluated the growth and survival of different age classes of blue oak seedlings in the foothills of Kern County, California. They found that older blue oak seedlings had higher rates of survival than younger seedlings. Their seedling studies demonstrated that up to 18% of blue oak seedlings within the site were more than 26 years old, and seedlings did not growth in height during a four year drought period that took place during the study. They also found that young saplings had higher rates of mortality, but if these young seedlings survive, they grow at a faster rate than older seedlings. Koenig and Knops (2007) found comparatively slow growth in a set of acorns planted in 1985, further supporting Phillips et al. (2007). This suggests that once a seedling survives and becomes established, there is greater chance that it will remain alive and recruit into a sapling, relative to a newly germinated seedling. In addition, these results point out the strong control of environmental conditions in growth variability and the sensitivity of regeneration to changing environmental conditions

It has been suggested that the introduction of annual grasses has been a major factor contributing to suppression of blue oak seedling growth and regeneration (e.g. Gordon and Rice 2000, Cheng and Bledsoe 2004). In a study that evaluated the effects of soil water between annual plants and blue oak seedlings, Gordon et al. (1989) and Gordon and Rice (2000) demonstrated that annual plant neighborhoods can influence the local availability of soil water for oak seedlings. They found that water content decreased most rapidly at high densities of *Bromus* sp and *Erodium* sp and this in turn impacted seedling root growth. In addition, they found that variation in early growth may affect the probability of seedling regeneration into sapling and larger size classes. Higher growth rates that persist longer into the summer drought would produce cumulatively larger seedlings with deeper roots, better able to exploit soil resources not available to the herbaceous understory (Gordon and Rice 2000).

Their results suggest that competition for soil water with introduced annual species contributes to the increased rate of blue oak seedling mortality observed in California. These studies suggest that the regeneration pattern of blue oak seedlings in California grasslands and woodlands may result from shifts in the seasonal soil water availability coincident with the replacement of native perennial herbaceous plants by Mediterranean annuals.

Conclusions and research needs.

In conclusion, these studies have identified that blue oak regeneration is occurring at SEKI, although it is occurring at a rate that does not compensate for adult mortality. Although fire has been identified as the major factor correlated with past regeneration peaks, concurrent changes in grazing intensity, fire, and arrival of invasive species limits our ability to rank the importance of these potential stressors in explaining current lack of regeneration. Beyond these three major suspects, other factors have been identified as major players in explaining temporal and spatial variability of regeneration. Acorn production is a time sensitive phenomenon that introduces large temporal variability in probability of successful regeneration. Similarly, sensitivity of seedling and saplings to environmental conditions not only influences when regeneration is most likely to occur, but also where. At smaller scales, sapling regeneration is higher under less dense canopy conditions, and at larger scales, elevation gradient and local climatic conditions limit sapling regeneration.

We recommend the following:

1. More research in less accessible areas, with particular emphasis to factors affecting blue oak growth that are applicable in SEKI.
2. Actively managing blue oak populations for grazing and impact of invasive species to ensure a seedling bank that guarantees future adult recruitment.
3. Given the spatial variability of factors influencing seedling and sapling growth, we recommend that any management practices incorporates an extensive representation of blue oak suitable habitats, rather than an intense protection of few representative areas.

Potential Stressors

Foothills at the Sequoia National Park are subject to several stressors due to proximity to progressively developed areas, its history of management, and the current regional and global trends in climatic conditions. We have shown in this chapter that characteristics of herbaceous, shrub, and tree communities at the foothills strongly vary with topographic and climatic conditions and specific characteristics at the watershed level. On top of strong differences between Sequoia and Kings Canyon units, native species tend to increase in diversity at higher elevations, where temperatures are lower and water availability during the growing season is greater. This relationship between diversity and elevation suggests that native plants might be strongly influenced by the global trend of increasing temperatures (Houghton et al. 2001), which is projected to be an increase of up to 3.8°C for the Sierra Nevada region (Snyder et al. 2002). This increase in temperatures will most likely imply an alteration of growing season conditions, in terms of length and water availability, as snowmelt runoff starts earlier in the season, reducing water availability in the summer (Cayan et al. 2001; Dettinger et al. 2004). Under these circumstances, native species might experience stronger competitive pressure from non-native annuals, especially with increased fire frequencies, and their ability to persist will depend on their ability to establish at higher elevations.

Because foothill communities at Sequoia National Park are protected, they have not experienced the same dramatic pressures as other foothill areas in the region. However, it is exceedingly difficult to establish robust reference conditions for this area, particularly with regard to fire regimes, and invasion of non-native annual grasses has occurred in the park, likely to a degree similar to outside the park. Given uncertainty in reference conditions, some evidence indicates that foothills may be diverging from their natural states due to changed disturbance regimes but more work is needed to set robust baseline conditions. In oak woodlands, for instance, fire suppression is thought to be responsible for the increase of non-native grasses, which was estimated to constitute as much as 99% of the herbaceous community in the foothills in the park (Parsons and Stohlgren 1989). In denser oak forests and shrublands, fire suppression may have reduced the advance of non-invasive grasses, but at the expense of an increased risk of severe fire. Despite this general expectation, we found that disturbance (based on vegetation cover change from Landsat analysis; Kennedy 2010) commonly occurred throughout the foothill area in the last several decades. While these disturbances appear to negatively affect some aspects of shrub diversity and native grass abundance, the levels do not appear strongly detrimental to many characteristics of the plant communities.

Increases in disturbance frequency, mainly due to more frequent fire, will likely contribute to an increase in the presence of non-native grasses at higher elevations when temperatures increase. Although we did not detect a strong influence of time since fire or fire return interval departures with most of our community metrics, these measures are highly uncertain and we did detect a decrease in native grass abundance with magnitude of disturbance based on landsat imagery. This decline is likely related to the effect of disturbance severity on seedbank composition and the mortality of adult perennial grasses. Severe disturbances that cause removal of at least 90% of the vegetation cover are associated with intense fires, which greatly reduce abundance and diversity of seedbanks (Keeley et al. 2003). Although these results did not allow us to evaluate foothill community responses to different disturbance frequencies, they do suggest increased

disturbance frequencies may be detrimental to some components of the native vegetation. If predicted fire frequency increases above limits that allow community recovery, which has been estimated at two fires within ten years, the spread of invasive species will be favored (Keeley 2006) and foothill shrublands will most likely convert to grasslands dominated by invasive species.

Air quality in the foothills of the Sequoia National Park has deteriorated in the past years mainly because of the constant accumulation of air pollutants in the San Joaquin Valley, which are trapped by thermal inversions during the night and are delivered to the western slope of the Sierras during the day (Cahill et al. 1996). Probably one of the most important trends in pollutants affecting community characteristics in the foothills is the increase in atmospheric nitrogen deposition. Nitrogen deposition along the SEKI foothills is considered higher than the critical threshold for observing changes in lichen communities in oak woodlands and chaparral, and increased dominance of invasive grasses (5.5 kg N/ha, Fenn et al. 2010). Higher nitrogen deposition is a well-documented factor favoring proliferation of invasive species, with the subsequent reduction in herb species diversity and increase in dominance in forest (e.g. Gilliam 2007, Allen et al. 2007). Although ozone levels have been on the rise in the San Joaquin Valley, symptoms of ozone exposure have not been detected in herbs and shrubs at the foothills (Campbell et al. 2000) or have been shown moderate levels even for ozone sensitive species. Moderate ozone induced injuries have been detected in some species present in SEKI (*Cayophytum diffusum*, *Lepidium virginicum*, *Cordylanthus rigidus*), like aboveground biomass reduction in ozone sensitive species like the native bunchgrass (*Elymus glaucus*, Temple 1999, Yoshida et al. 2001). In the light of our results, probably the most threatening impact of increased air pollution on characteristics of the foothills is the potential loss of the spatial variability detected in this study; this may start with increased presence of invasive species at higher elevations and be followed by spatial homogenization across watersheds.

In recent years, plant pathogens have been shown to be an underestimated risk for plant communities, in particular for foothills communities. Sudden oak death is the current most evident pathogenic threat for oak woodlands and hardwood forests. This disease is caused by the pathogen *Phytophthora ramorum*, which lethally infects its hosts, or causes non-lethal foliar and twig infections (Rizzo and Garbelotto 2003). A recent state-wide evaluation of the risk of establishment and spread of sudden oak death in California classified SEKI as at low risk (Meetenmeyer et al. 2004, Barrett and Waddell 2006). This classification is based on spread predictions calculated using climatic and topographic information, as well as the abundance and distribution of potential hosts. Blue oak is not known to be host of the pathogen, and known hosts that are dominant in SEKI foothills scored the lowest in terms of their potential to spread *P. ramorum* (species evaluated included live and California black oaks, California buckeye, and whiteleaf manzanita -*Artostaphylos Manzanita*). However, monitoring should be employed to ensure early detection of sudden oak death or a similar pathogen in SEKI; in particular, Gold Spotted Oak Borer (*Agilus auroguttatus*, GSOB). GSOB is potential threat SEKI foothill vegetation given that two of its host species are particularly abundant at SEKI (canyon live oak and black oak). Although the threat is not imminent, mortality in canyon live and black oak populations due to the attack of this beetle has been already detected in southern California (Coleman et al. 2011); for this reason we recommend a close monitoring of regional movements of this pest.

Assessment and Level of Confidence

In this chapter we have explored the relative conditions of plant communities in the foothill zone, with emphasis on community metrics across functional groups (herbaceous species, shrubs, and trees), and across vegetation types (shrublands and hardwood forests). Although our ability to detect the impact of specific potential stressors is weak, most of the variation we detected appears to be a consequence of differences across watersheds and strong trends in topographic and climatic factors, rather than a consequence of varying impacts of different potential stressors. Vegetation cover decline based on Landsat imagery was our most robust measure of disturbance; here we detected an impact on native grass abundance and little effect on most other diversity metrics. Although general response to levels of potential stressors does not indicate that foothill communities are at imminent risk, we have identified areas that are at a critical stage relative to other areas in the park. Table 11 summarizes our general assessment for each specific metric and identifies percentage of the foothill zone that is below our plot level estimates. Figures 16 and 17 identify critical areas at the scale of the watershed, indicating the level of confidence in our assessment. In general, our confidence levels are a direct consequence of the density and distribution of plots used during our analyses. Given that confidence in these predictions increases as diversity values increase for most of our metrics, we are more confident of low native herb diversity values than in high values; which is particularly true for Kings Canyon National Park where our confidence is relatively low.

Table 11. General conditions of foothill plant communities in relation to plot level estimates. For each of the metrics used, we estimated the percentage of the total foothill zone that is below the average calculated using NRI plots. This procedure highlights the overall spatial variation of conditions in the park.

Vegetation Type	Layer	Metric	Integrity Measure (average from plot measurements)	Condition (% of total foothill area below average)	Summary of Comments
Shrublands	Herbs	Shannon (H')	1.97 (SE=0.136 n=33)	Poor 96.2%	Lowest in Middle and East Fork Kaweah; Highest in North and Marble Fork Kaweah
		Abundance of Native Grasses	0.002 (SE=0.0007 n=33)	Poor 72.3%	Overall low abundance of native grasses, with highest values Marble Fork Kaweah and lowest at East Fork Kaweah
	Shrubs	Shannon (H')	0.82 (SE=0.10 n=32)	Poor 95.8%	Lowest in North and Marble Fork Kaweah.
Hardwood forest	Herbs	Shannon (H')	2.2 (SE=0.105 n=45)	Moderate 62.5%	Although the highest diversity of herbs was found in Hardwood forest, it is relatively high in Kings Canyon. Sequoia National Park showed the lowest diversity values
		Abundance of Native Grasses	0.008 (SE=0.003 n=45)	Poor 53%	Overall low abundance of native grasses, with highest values in Kings Canyon and Marble Fork unit
	Shrubs	Shannon (H')	1.07 (SE=0.10 n=35)	Poor 57.4%	Overall low with lowest in North and Marble Fork Kaweah.
	Trees	Shannon (H')	0.526 (SE=0.07 n=36)	Moderate 71.3%	Highest in Kings Canyon and East Fork Kaweah, strong increasing trend with elevation

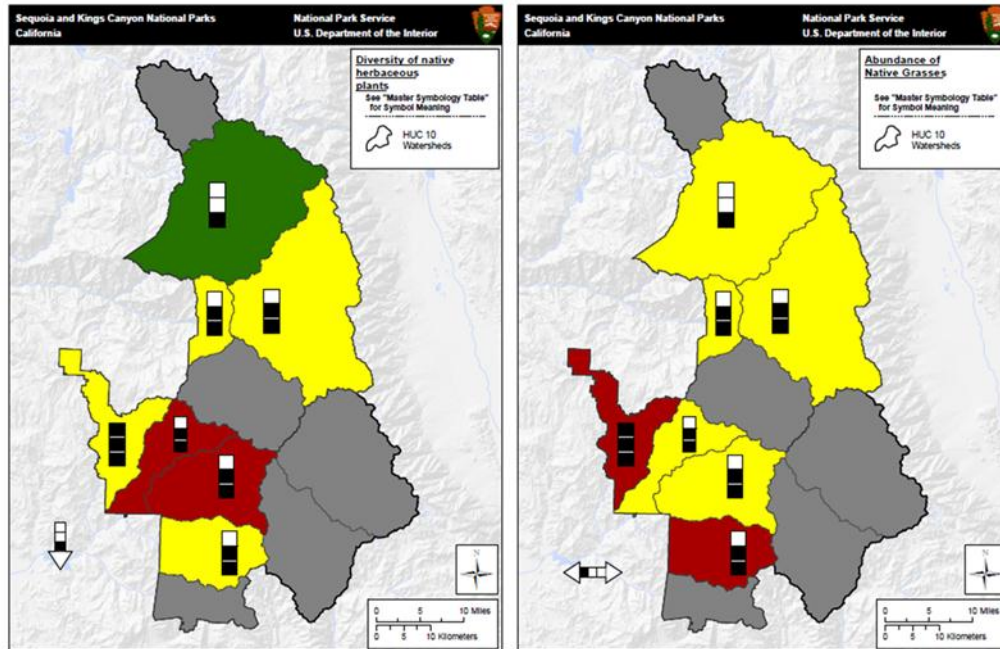


Figure 16. Assessment of current conditions and trends in native herbaceous communities throughout SEKI foothills. Estimation of current conditions is based on all our results and location of NRI plots; condition is reported for the entire watershed. Panel on the left indicates diversity of native herbs, panel on the right abundance of native grasses. The arrow (below right) summarizes the results of our trend analysis for the whole park, but in particular for the Sequoia National park where most of the plots were established

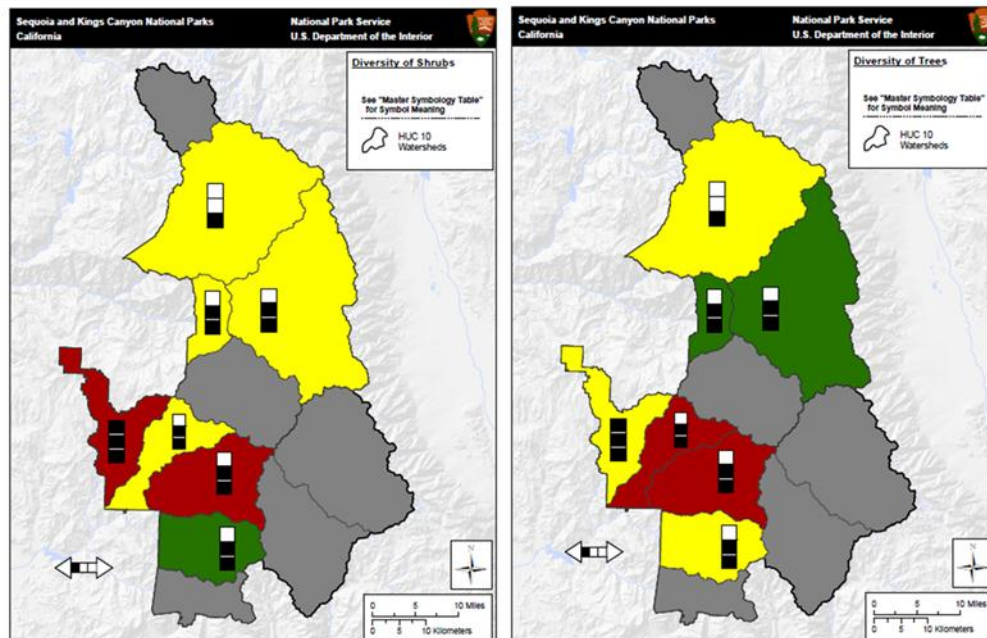


Figure 17. Assessment of current conditions and trends in shrub and tree communities throughout SEKI foothills. Estimation of current conditions is based on all our results and location of NRI plots; condition is reported for the entire watershed. Panel on the left indicates diversity of shrubs and panel on the right diversity of trees. The arrow summarized the results of our trend analysis for the whole park, but in particular for the Sequoia National park where most of the plots were established

Gaps in Understanding

Our emphasis in this chapter has been exploring the spatial variability of community diversity metrics throughout the foothills. We have tried to disentangle importance of factors that define environmental conditions, and factors that summarize history of disturbance (vegetation cover change as a general measure, as well as fire return interval departures, and grazing history). In general, we have a very weak understanding of the role of potential stressors in the foothills; more study on fire and grazing as well as other potential stressors such as nitrogen deposition are needed. In addition, without a clear understanding of the mechanisms that govern community assembly in the foothills, we are unable to predict the impact of future conditions at the community level. For example, current knowledge of the mechanisms that govern postfire community composition in shrublands (Keeley and Fotheringham 2003, Keeley et al. 2005b) allowed us to generate a hypothesis of the effect of increasing fire return intervals or disturbance, but this knowledge is not generalized for oak forests, in particular in areas where blue oak is not dominant. Communities dominated by canyon live oak have been less studied, and although they are considered more stable (Bolsinger 1988), there is a general lack of understanding of mechanisms governing response of these communities to disturbance, fire, or any of the other potential stressors acting in the park. In addition, there is very little knowledge of the status and distribution of native grasses in the foothill region; this vegetation type is in need of more research within the park.

The lack of mechanistic understanding was particularly limiting when we reviewed the state of knowledge regarding regeneration in oak populations. Demographic modeling is a key missing component concerning oak regeneration; efforts along these lines are needed to determine if oak populations are declining and, if so, what the most vulnerable life history stages are. In addition, although blue oak is the most prominent species in oak woodlands, canyon live oak assemblages are the dominant communities in SEKI foothills. There is a plethora of information about blue oak response to fire, grazing, and climatic changes, but information regarding other species is scarce. In order to fully assess the status of foothill vegetation at SEKI, future research should prioritize areas not dominated by blue oak, and explicitly explore questions regarding community assembly mechanisms in these areas.

Recommendations for Future Study/Research

During our analyses, we have identified several aspects that are poorly understood and that are a priority to accurately assess the current conditions of foothill plant communities:

- Our assessment is constrained by the distribution of sampled areas with respect to the full extent of the foothills. Several areas in the foothills are not represented by the NRI plots (we base our analysis on these plots, as they are the best available despite the limited sampling extent) and the more extensive dataset, VMA plots, have concerns about the biased location of the plots. It is crucial to determine how representative the NRI plots (or a combination of other monitoring plots) are with respect to spatial patterns across the entire foothill region of SEKI. This validation is particularly relevant native herbaceous

species, especially native grasses, given that our predictions estimate high diversity for this guild in areas that have not been sampled.

- We had limited information to assess temporal trends. We recommend recensusing plots on a regular basis. This is particularly relevant because our analyses are based on changes in spatial patterns rather than absolute changes in community metrics. For instance, it will be important to understand whether increased abundance of invasive grasses is associated with declines in native diversity.
- In addition to overall monitoring, we recommend focused monitoring and research program in two vegetation types of high conservation concern in the foothill area: blue oak woodlands and native perennial grasslands. It is difficult to use the limited monitoring data available to assess conditions or trends in these vegetation types, yet their status should be of high concern to SEKI. In particular, demographic studies of oak population and distribution mapping of native grasses are high priority research areas.
- While we can speculate on the role of potential stressors (e.g., fire, grazing, invasive plants), there is very limited understanding about fire or grazing history in the foothill areas of SEKI. For instance FRID values for low elevation areas are highly uncertain or unknown due to the difficulties in assessing historical fire regimes in these areas. This strongly limits our ability to assess current conditions relative to these potential stressors.

Literature Cited

- Alagona, P. S. 2008. Homes on the range: cooperative conservation and environmental change on California's privately owned hardwood rangelands. *Environmental History* 13: 325-349.
- Allen, E. B., P. J. Temple, A. Bytnerowicz, M. J. Arbaugh, A. G. Sirulnik, and L. E. Rao. 2007. Patterns of understory diversity in mixed coniferous forests of southern California impacted by air pollution. *Scientific World Journal* 7: 247-263.
- Allen-Diaz B. H. and J. W. Bartolome. 1992. Survival of *Quercus douglasii* (Fagaceae) seedlings under the influence of fire and grazing. *Madroño* 39:47-53.
- Allen-Diaz, B. H. and B. A. Holzman. 1991. Blue oak communities in California. *Madroño* 38: 80-95.
- Allen-Diaz, B. H. and R. D. Jackson. 2005. Herbaceous responses to livestock grazing in californian oak woodlands: A review for habitat improvement and conservation potential. USDA Forest Service Gen. Tech. Rep. PSW-GTR-195.
- Arevalo, J. R., P. Alvarez, N. Narvaez, and K. Walker. 2009. The effects of fire on the regeneration of a *Quercus douglasii* stand in Quail Ridge Reserve, Berryessa Valley (California). *Journal of Forest Research* 14:81-87.
- Barrett, T. and K. Waddell. 2006. Regeneration of California oak woodlands 2001-2005. USDA Forest Service General Technical Report PSW-GTR-217.
- Barrett, T. M., D. Gatzolis, J. S. Fried, and K. L. Waddell. 2006. Sudden oak death in California: What is the potential? *Journal of Forestry* 104:61-64.
- Bartolome, J. W., P. C. Muick, and M. P. McClaren. 1987. Natural regeneration of Californian hardwoods. In: Proc. symp. on multiple-use management of California's hardwood resources; 1986 San Luis Obispo, Calif. USDA Forest Service, Pacific Southwest Forest and Range Exp. Station, Berkeley, California. Gen Tech. Rep. PSW-100:26- 31.
- Bolsinger, C. L. 1988. The hardwoods of California's timberlands, woodlands, and savannas. Resource Bulletin PNW-RB-148. Portland (OR): Pacific Northwest Research Station, Forest Service, U.S. Department of Agriculture.
- Borchert, M. I., F. W. Davis, J. Michaelsen, and L. D. Oyler. 1989. Interactions of factors affecting seedling regeneration of blue oak (*Quercus douglasii*) in California. *Ecology* 70:389-404.
- Cahill, T. A., J. J. Carroll, D. Campbbell, and T. E. Gill. 1996. Air Quality. Sierra Nevada Ecosystem Project: final report to Congress. Centers for Water and Wildlands Resources, editor. UC Davis, Davis, CA.
- Campbell S., G. Smith, P. Temple, J. Pronos, R. Rochefort, and C. Andersen C. 2000. Monitoring for ozone injury in West Coast (Oregon, Washington, California) forests in

1998. Portland (OR): USDA, Forest Service, Pacific Northwest Research Station. Gen Tech Report PNW-GTR-495. 19 p.
- Caprio, A. C. and P. Lineback. 2002. Pre-Twentieth Century Fire History of Sequoia and Kings Canyon National Parks: A Review and Evaluation of Our Knowledge. pp 180-199. In: Proceedings of the Symposium: Fire in California Ecosystems: Integrating Ecology, Prevention and Management, San Diego, California, Nov. 17-20, 1997. N.G. Sugihara, M.E. Morales, and T.J. Morales (eds). 2002. Misc. Pub. No. 1, Assoc. for Fire Ecology. 391 pp.
- Cayan, D. R., S. A. Kammerdiener, M. D. Dettinger, J. M. Caprio, and D. H. Peterson. 2001. Changes in the onset of spring in the western United States. *Bulletin of the American Meteorological Society* 82:399-415.
- Cheng, X. M. and C. S. Bledsoe. 2004. Competition for inorganic and organic N by blue oak (*Quercus douglasii*) seedlings, an annual grass, and soil microorganisms in a pot study. *Soil Biology & Biochemistry* 36:135-144.
- Clements, F. E. 1920. Plant indicators. Carnegie Institute of Washington Publications 290: 1–388.
- Coleman, T. W., N. E. Grulke, M. Daly, C. Godinez, S. L. Schilling, P.J. Riggan, S. J. Seybold. 2011. Coast live oak, *Quercus agrifolia*, susceptibility and response to goldspotted oak borer, *Agrilus auroguttatus*, injury in southern California. *Forest Ecology and Management* 261:1852–1865.
- Conard, S. G. and D. R. Weise. 1998. Management of fire regime, fuels, and fire effects in southern California chaparral: lessons from the past and thoughts for the future. In: T. L. Pruden and L. A. Brennan (eds.): Fire in ecosystem management: shifting the paradigm from suppression to prescription. Tall Timbers Fire Ecology Conference Proceedings, No. 20. Tall Timbers Research Station, Tallahassee, FL; pp: 342–350.
- Davis, F. W., R. Brown, B. Buyan, J. Kealy. 1995. Vegetation change in blue oak and blue oak/foothill pine woodland. A report to the California Department of Forestry and Fire Protection, Sacramento (CA).
- Dettinger, M. D., D. R. Cayan, M. Meyer, and A. E. Jeton. 2004. Simulated hydrologic responses to climate variations and change in the Merced, Carson, and American River basins, Sierra Nevada, California, 1900-2099. *Climatic Change* 62:283-317.
- DiTomaso, J. M., G. B. Kyser, M. R. George, M.P. Doran, and E.A. Laca. 2008. Control of medusahead (*Taeniatherum caput-medusae*) using timely sheep grazing. *Invasive Plant Science and Management* 1:241–247.
- Fenn M.E., E.B. Allen, S.B. Weiss, S. Jovan, L.H. Geiser, G.S. Tonnesen, R.F. Johnson, L.E. Rao, B.S. Gimeno, F. Yuan, T. Meixner, and A. Bytnerowicz. 2010. Nitrogen critical loads and management alternatives for N-impacted ecosystems in California. *Journal of Environmental Management* 91:2404–2423.

- Gilliam, F. S. 2007. The ecological significance of the herbaceous layer in forest ecosystems. *BioScience* 57:845–858.
- Gordon, D. R. and K. J. Rice. 1993. Competitive effects of grassland annuals on soil-water and blue oak (*Quercus douglasii*) seedlings. *Ecology* 74:68-82.
- Gordon, D. R. and K. J. Rice. 2000. Competitive suppression of *Quercus douglasii* (Fagaceae) seedling emergence and growth. *American Journal of Botany* 87:986-994.
- Gordon, D. R., J. M. Welker, J. W. Menke, K. J. Rice. 1989. Competition of soil water between annual plants and blue oak (*Quercus douglasii*) seedlings. *Oecologia* (Berlin) 79:533–541.
- Griffin, J.R. 1973 Valley oaks –and end of an era? *Fremontia*, 1:5-9.
- Haggerty, P. K. 1994. Damage and recovery in southern Sierra Nevada foothill oak woodland after a severe ground fire. *Madroño*, 41: 185–198.
- Hall, L. M., M. R. George, D. D. McCreary, and T. E. Adams. 1992. Effects of cattle grazing on blue oak seedling damage and survival. *Journal of Range Management* 45:503-506.
- Hamilton, J.G. 1997. Changing perceptions of pre-European grasslands in California. *Madroño* 44:311-333.
- Harvey L E. 1989. Spatial and temporal dynamics of a blue oak woodland. PhD dissertation, University of California, Santa Barbara.
- Hayes, G. F. and K. D. Holl. 2003. Cattle grazing impacts on annual forbs and vegetation composition of mesic grasslands in California. *Conservation Biology*, 17:1694–1702.
- Heady, H. F. 1977. Valley grasslands. In: Barbour, M.G. and J. Major (eds.): *Terrestrial Vegetation of California*. John Wiley, New York, NY. pp 491-514.
- Holland, V. L. and D. J. Keil. 1995. *California Vegetation*. Kendall/Hunt Publishing Co., Dubuque, IA.
- Holstein, G. 2001. Pre-agricultural grassland in central California. *Madroño* 48: 253–264.
- Holzman, B. A. 1993. Vegetation change in California's blue oak (*Quercus douglasii*) woodlands 1932–1992. Ph.D. dissertation, University of California, Berkeley.
- Hopkinson, P., and L. Huntsinger. 2005. Are East Bay hills grasslands a historical artifact? Phytolith evidence and a potential candidate for the true East Bay vegetation type. *Grasslands* 15:7-9.
- Houghton, J. T., D. J. Ding, M. Griggs, M. Noguer, P. J. van der Linden, X. Dai, K. Maskell, and C. A. Johnson. 2001. *Climate Change 2001: The Scientific Basis*. Cambridge University Press, Cambridge, U.K.

- Huntsinger, L., J.W. Bartolome, and C. M. D'Antonio. 2007. Grazing management on California's Mediterranean grasslands. In: Stromberg, M.R., J.D. Corbin and C.M. D'Antonio (eds.): California Grasslands. University of California Press, Berkeley, CA. pp 233-253.
- Jackson, R. D., B. Allen-Diaz. 2006. Grazing disturbance promotes equilibrium plant community dynamics in an inherently non-equilibrium system. *Journal of Applied Ecology* 43:485-496.
- Johnson, P. S. 1977. Predicting oak stump sprouting and sprout development in the Missouri Ozarks. USDA Forest Service Research Paper, North Central Forest Experiment Station:11 pp.
- Johnson, R. A. and D.W. Wichern. 2007. *Applied multivariate statistical analysis*. Pearson Prentice Hall (6th edition). New Jersey, U.S.A.
- Keeley, J. E. 1986. Resilience of mediterranean shrub communities to fires. In: B. Dell, A.J.M. Hopkins, and B.B. Lamont, editors. Resilience in Mediterranean-type ecosystems. Dr. W. Junk Publishers, Dordrecht, Netherlands; pp: 95–112.
- Keeley, J.E. 1990. The California Valley Grassland. In: A.A. Schoenherr (editor). Endangered plant communities of Southern California. Special Publication No. 3, Southern California Botanists, Fullerton, CA. pp 2-23.
- Keeley, J. E. 1993. Native grassland restoration: the initial stage — assessing suitable sites. In J.E. Keeley (editor), Interface Between Ecology and Land Development in California. Southern California Academy of Sciences, Los Angeles, CA. .
- Keeley, J. E., D. Lubin, and C. J. Fotheringham. 2003. Fire and grazing impacts on plant diversity and alien plant invasions in the southern Sierra Nevada. *Ecological Applications* 13:1355-1374.
- Keeley, J. E. and C.J. Fotheringham. 2003. Species–area relationships in Mediterranean-climate plant communities. *Journal of Biogeography*, 30: 1629–1657.
- Keeley, J. E., M. Baer-Keeley, and C. J. Fotheringham. 2005a. Alien plant dynamics following fire in Mediterranean-climate California shrublands. *Ecological Applications* 15:2109-2125.
- Keeley, J. E., A.H. Pfaff, and H.D. Stafford. 2005b. Fire suppression impacts on postfire recovery of Sierra Nevada chaparral shrublands. *International Journal of Wildland Fire* 14: 255–265.
- Keeley, J.E., T. Brennan, and A.H. Pfaff. 2008. Fire severity and ecosystem responses following crown fires in California shrublands. *Ecological Applications* 18:1530-1546.
- Kennedy, R. 2010. Landsat-based monitoring of landscape dynamics in the national parks of the Sierra Nevada Inventory & Monitoring Network” (SIEN). Oregon State University.

- Koenig, W. Mumme, R. Carmen, W. and Stanback, M. 1994. "Acorn production by oaks in Central Coastal California: variation within and among years." *Ecology*. 75(1) 99-109.
- Koenig, W. D., J. M. H. Knops, W. J. Carmen, and M. T. Stanback. 1999. Spatial dynamics in the absence of dispersal: acorn production by oaks in central coastal California. *Ecography* 22: 499-506.
- Koenig, W.D. and J.M. Knops. 2007. Long-term growth and persistence of blue oak (*Quercus douglasii*) seedlings in a California oak savanna. *Madroño* 54: 269-274.
- Kuhn, B. and B. Johnson. 2008. Status and trends of black oak (*Quercus kelloggii*) populations and recruitment in Yosemite Valley (a.k.a. Preserving Yosemite's oaks). Final report to Yosemite Fund. July, 2008, 50 pp.
- Lozano F. J., S. Suarez-Seoane, and E. Luis. 2010. Effects of wildfires on environmental variability: a comparative analysis using different spectral indices, patch metrics and thematic resolutions. *Landscape Ecology* 25:697-710.
- Lozano F. J., S. Suarez-Seoane, and E. Luis. 2010. Effects of wildfires on environmental variability: a comparative analysis using different spectral indices, patch metrics and thematic resolutions. *Landscape Ecology* 25:697-710.
- Marty, J. T. 2005. Effects of cattle grazing on diversity in ephemeral wetlands. *Conservation Biology*, 19: 1626-1632.
- McClaran, M. P. 1985. Regeneration of California blue oak as determined from age structures at multiple bole heights. *Bulletin of the Ecological Society of America* 66:226-226.
- McClaran, M. 1986. Age structure of *Quercus douglasii* in relation to livestock grazing and fire. PhD dissertation, University of California, Berkeley.
- McClaran, M. P. and J. W. Bartolome. 1989. Fire related regeneration in stagnant *Quercus douglasii* populations. *Canadian Journal of Forest Research/Revue Canadienne De Recherche Forestiere* 19:580-585.
- McCreary, D. D., and J. Tecklin. 1997. Effects of seedling protectors and weed control on blue oak growth and survival. Pp. 243-250 in Pillsbury et al (eds). Proceedings of a symposium on oak woodlands: ePSW-GTR-160.
- McCreary, D.D., B.Tietje, and B. Frost. 2006. Stump sprouting of Blue oaks 19 Years after harvest. USDA General technical report PSW-GTR-217, pp: 333-341.
- Meentemeyer R., D. Rizzo, W. Mark, and E. Lotz, 2004. Mapping the risk of establishment and spread of sudden oak death in California. *Forest Ecology and Management* 200: 195-214.
- Menke, J. W., C. Davis, and P. Beesley. 1996. Rangeland assessment. In: Sierra Nevada Ecosystem Project: Final Report to Congress. Status of the Sierra Nevada. Volume III. Centers for Water and Wildland Resources, University of California, Davis, California, USA. pp: 901-972.

- Mensing, S. A. 1988. blue oak (*Quercus douglasii*) regeneration in the Tehachapi Mountains, Kern County, California. Master's thesis, University of California, Berkeley.
- Mensing, S. A. 1992. The impact of European settlement on blue oak (*Quercus douglasii*) regeneration and regeneration in the Tehachapi Mountains, California. *Madroño* 39:36-46.
- Minnich, R. A. 1995. Fuel-driven fire regimes of the California chaparral. In: Keeley, J.E. and T. Scott (eds.): *Brushfires in California wildlands: ecology and resource management*. International Association of Wildland Fire, Fairfield, WA; 21–27.
- Muick, P. C., and J. W. Bartolome. 1987. Factors associated with oak regeneration in California. In: Plumb, T. R.; Pillsbury, N. H., (technical coordinators). *Proceedings of the symposium on multiple-use management of California's hardwood resources; 1986 November 12-14; San Luis Obispo, CA. General Technical Report. PSW-100*. Berkeley, CA: U.S: Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 86-91.
- Parker, V. T., and C. H. Muller. 1982. Vegetational and Environmental-Changes beneath Isolated Live Oak Trees (*Quercus-Agrifolia*) in a California Annual Grassland. *American Midland Naturalist* 107:69-81.
- Parsons, D. J., and T. J. Stohlgren. 1989. Effects of varying fire regimes on annual grasslands in the southern Sierra Nevada of California. *Madroño* 36:154–168.
- Parsons, D. J., P. W. Rundel, R. P. Hedlund, and G. A. Baker. 1981. Survival of severe drought by non-sprouting chaparral shrub. *American Journal of Botany* 68:973-979.
- Phillips, R. L. 1992. Environmental factors contribute to acorn quality...Elevation, on- or off-tree collection influence the viability of blue oak acorns. *California Agriculture* 46:30-32.
- Phillips, R. L., N. K. McDougald, D. McCreary, and E. R. Atwill. 2007. Blue oak seedling age influences growth and mortality. *California Agriculture* 61:11-15.
- Phillips RL, McDougald NK, Standiford RB, et al.1997. Blue oak regeneration in southern Sierra Nevada foothills. In: Pillsbury NH, Verner J, Tietje WD (tech. coords.). *Proc Symp on Oak Woodlands: Ecology, Management and Urban Interface Issues*. March 19–22, 1996, San Luis Obispo, CA. USDA Forest Service Gen Tech Rep PSW-160:177–81.
- Phillips, R. L., N. K. McDougald, R. B. Standiford, and W. E. Frost. 1996. Blue oak seedlings may be older than they look. *California Agriculture* 50:17-19.
- Rizzo D. M., and M. Garbelotto, 2003. Sudden oak death: endangering California and Oregon forest ecosystems. *Frontiers in Ecology and the Environment* 1: 197–204.
- Roy, D. G. and J. L. Vankat. 1999. Reversal of human-induced vegetation changes in Sequoia National Park, California. *Canadian Journal of Forest Research/Revue Canadienne De Recherche Forestiere* 29:399-412.

- Schiffman, P. M. 2007. Ecology of native animals in California grasslands. pp. 180-190 in Stromberg, M.R., J.D. Corbin, and C.M. D'Antonio (eds.) *California Grasslands: Ecology and Management*. University of California Press, Berkeley.
- Schmidt, R. H. and R. M. Timm. 1991. Vertebrate impacts on oak regeneration in California: a review of management options. *Proceedings Great Plains Wildlife Damage Prevention and Control Workshop* 10:134-144.
- Snyder, M. A., J. L. Bell, L. C. Sloan, P. B. Duffy, and B. Govindasamy. 2002. Climate responses to a doubling of atmospheric carbon dioxide for a climatically vulnerable region. *Geophysical Research Letters* 29: 1-4.
- Solomeschch, A. I. and M. G. Barbour 2006. Defining restoration targets for the Great Valley Grassland State Park (San Joaquin Valley). *Grasslands* 16:1-17.
- Stahle, D. W., R. D. Griffin, M. K. Cleaveland, J. R. Edmondson, F. K. Fye, D. J. Burnette, J. T. Abatzoglou, K. T. Redmond, D. M. Meko, M. D. Dettinger, and D. R. Cayan. 2010. Hydroclimatic reconstruction and ancient blue oak mapping over the drainage basin of san francisco bay. Final report. CALFED Ecosystem Restoration Program, Project ERP-02-P30. 291pp.
- Stahle, D. W., R. D. Griffin, M. D. Therrell, J. R. Edmondson, M.K. Cleaveland. L. N. Stahle, D. J. Burnette, J. T. Abatzoglou, K. T. Redmond, D. M. Meko, M. D. Dettinger, and D.R. Cayan, 2013. The ancient blue oak woodlands of California. *Earth Interactions* (in press).
- Standiford, R.B, N.K. McDougald, R.L. Phillips, and A. Nelson. 1991. South Sierra oak regeneration weak in sapling stage. *California Agriculture* 45: 12-14
- Standiford, R., B. McDougald, W. Frost, and R.L. Phillips. 1997. Factors influencing the probability of oak regeneration on southern Sierra Nevada woodlands in California. *Madrone* 44:170-183.
- Standiford, R. B., J. Vreeland, and B. Tietje. 2001. *California's Hardwood Rangelands: Production and Conservation Values*. Berkley, California: University of California Integrated Hardwood Range Management Program, UC Berkeley.
- Stephens, S. L. 1998. Evaluation of the effects of silvicultural and fuels treatments on potential fire behavior in Sierra Nevada mixed-conifer forests. *Forest Ecology and Management* 105:21-35.
- Sudworth, G. B. 1908. *Forest Trees of the Pacific Slope*. Washington (DC): Government Printing Office.
- Swiecki T. J., E. A. Bernhardt, and C. Drake. 1993. Factors affecting blue oak sapling recruitment and regeneration. A report to the California Department of Forestry and Fire Protection, Strategic Planning Program, Sacramento (CA). Contract 8 CA17358.

- Swiecki, T. J., E. A. Bernhardt, and C. Drake. 1997a. Stand-level status of blue oak sapling recruitment and regeneration. USDA Forest Service General Technical Report PSWPSW-160, pp.147–156.
- Swiecki, T. J., E. A. Bernhardt, and C. Drake. 1997b. Factors affecting blue oak sapling regeneration. USDA Forest Service General Technical Report PSWPSW-160, pp.157–167.
- Swiecki, T. J. and E. A. Bernhardt. 1998. Understanding blue oak regeneration. *Fremontia* **26**: 19-26.
- Swiecki, T. J. and E. A. Bernhardt. 2001. Effects of fire on naturally occurring Blue oak (*Quercus douglassii*) saplings. USDA Forest Service General Technical Report PSWPSW-184, pp.251–239.
- Tecklin, J., and D. McCreary. 1991. Acorn size as a factor in early seedling growth of blue oak. USDA Forest Service General Technical Report PSW-126: 48–53.
- Temple, P. J., 1999. Effects of ozone on understory vegetation in the mixed conifer forest. In: Miller, P.R., McBride, J.R. (eds.): Oxidant Air Pollution Impacts on the Montane Forests of Southern California: A Case Study of the San Bernardino Mountains, pp. 208–222.
- Tyler, C. M., B. Kuhn, and F. W. Davis. 2006. Demography and regeneration limitations of three oak species in California. *Quarterly Review of Biology* 81:127-152.
- Vankat, J. L. 1970. Vegetation changes in Sequoia National Park, California. Ph.D. dissertation, University of California, Davis.
- Vankat, J. L. 1977. Fire and man in Sequoia National Park. *Annals of the Association of American Geographers* 67:17-27.
- Vankat, J. L., and Major, J. 1978. Vegetation changes in Sequoia National Park, California. *Journal of Biogeography* 5: 377–402.
- White, K. L. 1966. Structure and composition of foothill woodland in central-coastal California. *Ecology* 47:229-337
- Yoshida, L. C., J. A. Gamon, and C. P. Andersen. 2001. Differences in above-and below-ground responses to ozone between two populations of a perennial grass. *Plant and Soil* 233: 203–211.
- Zedler, P. H. 1995. Fire frequency in southern California shrublands: biological effects and management options. In: Keeley, J.E. and T. Scott, editors. Brushfires in California wildlands: ecology and resource management. International Association of Wildland Fire, Fairfield, WA; pp:101–112.

Appendix

Table A 1. Detailed description of vegetation communities at the foothills of the SEKI.

Alliance common name grouped by vegetation subclasses	% Total Foothills Area	Average elevation (m a.s.l.)
Shrubland		
Deciduous shrubland		
Birchleaf Mountain Mahogany Shrubland Alliance	2.1%	1242.46
Birchleaf Mountain Mahogany-California Redbud-California Flannelbush Shrubland Association	1.3%	986.51
Birchleaf Mountain Mahogany-Whiteleaf Manzanita Shrubland Association	2.0%	1410.83
Bitter Cherry Shrubland Alliance	0.02%	1767.67
Bitter Cherry-Gooseberry spp. (Mountain Maple) Shrubland Mapping Unit	0.09%	1800.05
California Grape Association	0.01%	902.28
Deerbrush Shrubland Alliance	1.1%	1662.41
Oregon White Oak Shrubland Alliance	0.31%	1279.24
Oregon White Oak-Birchleaf Mountain Mahogany Shrubland Association	3.1%	1344.71
Willow spp. Riparian Shrubland Mapping Unit	0.19%	1644.75
Willow spp./Meadow Shrubland Mapping Unit	0.05%	1732.67
Deciduous shrubland Total	10%	1364.27
Evergreen shrubland		
Big Sagebrush Shrubland Alliance	0.04%	1812.11
Buckbrush Shrubland Alliance	0.14%	1282.22
Chamise Shrubland Alliance	7.7%	989.85
Chamise-Buckbrush Shrubland Association	0.59%	1130.20
Chamise-California Yerba Santa Shrubland Association	2.1%	1029.77
Chamise-Chaparral Yucca Shrubland Association	0.21%	908.96
Chamise-Whiteleaf Manzanita Shrubland Association	32%	1114.86
Chaparral Whitethorn Shrubland Alliance	0.25%	1147.02
Chaparral Yucca Shrubland Alliance	0.60%	1139.89
Greenleaf Manzanita Shrubland Alliance	0.22%	1699.63
Greenleaf Manzanita-Bush Chinquapin-Whitethorn Ceanothus Shrubland Superalliance	0.21%	1760.01
Indian Manzanita Shrubland Alliance	0.85%	1570.63
Mountain Misery Dwarf-shrubland Alliance	0.14%	1728.99
Whiteleaf Manzanita Shrubland Alliance	0.70%	1241.58
Evergreen shrubland Total	16.9%	1163.91
Mixed shrub/herbaceous		
Post Fire Shrub/Herbaceous Mapping Unit	0.17%	1634.41
Mixed shrub/herbaceous Total	0.17%	1634.41
Shrubland Total	27%	1256.29

Table A 2. Detailed description of vegetation communities at the foothills in SEKI.

Alliance common name grouped by vegetation subclasses	% Total Foothills Area	Average elevation (m a.s.l.)
Hardwood Forest		
Deciduous forest		
Bigleaf Maple Forest Alliance	0.17%	1580.21
Black Cottonwood Forest Association	0.06%	1688.28
Black Cottonwood Temporarily Flooded Forest Alliance	0.27%	1574.47
Montane Broadleaf Deciduous Trees Mapping Unit	0.01%	1726.21
Quaking Aspen Forest Alliance	0.00%	1819.57
White Alder Temporarily Flooded Forest Alliance	0.54%	1421.58
White Alder-Red willow-California Sycamore Forest Association	0.16%	752.99
Deciduous forest Total	1.2%	1473.72
Deciduous woodland		
blue oak Woodland Alliance	0.01%	957.24
blue oak/Brome spp.- American Wild Carrot Woodland Association	0.81%	767.26
blue oak-California Buckeye-(Interior Live Oak) Woodland Mapping Unit	1.3%	821.81
blue oak-Interior Live Oak/Brome spp.- American Wild Carrot Woodland Association	1.3%	695.99
California Buckeye Woodland Alliance	0.11%	911.72
California Buckeye- Canyon Live Oak Woodland Association	1.9%	1210.90
California Sycamore Temporarily Flooded Woodland Alliance	0.03%	611.88
California Sycamore-(Canyon Live Oak- Interior Live Oak) Forest Mapping Unit	0.41%	697.95
Deciduous woodland Total	5.8%	904.51
Evergreen forest		
Canyon Live Oak Forest Alliance	6.5%	1548.68
Canyon Live Oak-(Ponderosa Pine- Incense-cedar) Forest Superassociation	2.6%	1435.58
Canyon Live Oak/Birchleaf Mountain Mahogany Forest Mapping Unit	3.2%	1379.83
Canyon live oak/Greenleaf Manzanita Forest Association	0.97%	1748.15
Canyon Live Oak/Whiteleaf Manzanita Forest Association	3.1%	1536.41
Canyon Live Oak-California Laurel Forest Superassociation	16%	1359.60
Interior Live Oak Woodland Alliance	1.3%	854.51
Interior Live Oak-California Buckeye/ Birchleaf Mountain Mahogany- California Redbud Forest Association	5.7%	875.06
Interior Live Oak- Canyon Live Oak Woodland Association	2.1%	997.56
Evergreen forest Total	42%	1310.31
Mixed evergreen deciduous forest	11.2%	1592.96
Hardwood Forest Total	60%	1329.48
Baren	4.0%	1473.78
Mixed Coniferous Forest	9.1%	1571.03
Grand Total	100%	1334.54

Table B. Distribution of study plots in SEKI. The number of plots at each watershed unit and the percentage of the dataset this represents are indicated above each watershed unit. Acronyms for each watershed unit are as follows: MFK=Middle Fork Kings River; USFK and LSKF are Upper and Lower South Fork Kings River, respectively; MbFKW, NFKW, MFKW, EFKW, SFKW correspond to Marble, North, Middle, East, and South Fork Kaweah River, respectively; NFT=North Fork Tule River.

DATASET													
Major Watershed	Watershed Code	Fire Effects		Gradient		NRI		Vankat-Roy		VM Accuracy		VM Full Plots	
		Number of plots	% Dataset	Number of plots	% Dataset	Number of plots	% Dataset	Number of plots	% Dataset	Number of plots	% Dataset	Number of plots	% Dataset
Kings River	MFK	0		0		3	3.9%	0		50	13%	12	10%
	USFK	0		0		6	7.8%	0		25	6.5%	20	17%
	LSFK	0		0		4	5.2%	0		18	4.7%	13	11%
	MbFKW	0		8	62%	16	21%	14	5	44	11%	34	29%
Kaweah River	NFKW	3	18%	1	7.7%	20	26%	2	7.1%	55	14%	15	13%
	MFKW	5	29%	3	23%	14	18%	12	43%	31	8%	10	8.6%
	EFKW	9	53%	1	7.7%	13	17%	0		130	34%	0	
	SFKW	0		0		1	1.3%	0		32	8.3%	12	10%
Tule River	NFT	0		0		0		0		2	0.52%	0	

Table C.. Dataset-specific manipulations during the estimation of community metrics. Fore each dataset we estimated relative abundance of each species grouped by vegetation stratum (herbs, shrubs, or trees).

	VMA	VM full plots	NRI	Gradient	Fire Effects	Vankat-Roy	
Herb	Samplin g	0.1Ha rectangular plot	NA	6 quadrants in groups of 2 along each of the four aspects of the plot (North, South, East, West)	25 small quadrants	0.1 Ha plots in forest and 30m strip in shrubland	NA
	Data	Cover per species	NA	% cover of each species at each quadrant	% cover class per species. Seven classes were used with the following midpoint (0.5%,3%,7.5%,17.5%,37.5%,62.5%,87.5%)	cover per species	NA
	Abunda nce Estimati ons	Sum cover per species divided total cover per plot per stratum	NA	To estimate species abundace per plot we add species abundance across plots	Sum midpoint cover per species divided by 2187.5	Sum cover per species divided total cover per plot per stratum	NA
Shrubs	Samplin g	0.1Ha rectangular plot	NA	2 perpendicular transect dissecting the plot in directions North and East. Total length of the transects varies	Line intercept method	0.1 Ha plots in forest and 30m strip in shrubland	NA
	Data	Cover per species	NA	Linear intercept of each species along	per species	cover per species	NA
	Abunda nce estimati ons	Sum cover per species divided total cover per plot per stratum	NA	Sum of point intercept divided by the total transect length	Sum of cover values per species divided by 2500	Sum cover per species divided total cover per plot per stratum	NA
Tree	Samplin g	0.1Ha rectangular plot	0.1 rectangular plots	0.1ha circular plots	0.1ha rectangular plots	0.1 Ha plots in forest and 30m strip in shrubland	0.01Ha transect
	Data	Cover per species	Diameter at breast height (DBH) trees taller than 1.4 m	Diameter at breast height (DBH) trees taller than 1.3 m	Class of Diameter at breast height (DBH) trees taller than 1.4 m. Classes of 5 cm width were used.	DBH per species	DBH per species
	Manipul ation	Sum cover per species divided total cover per plot per stratum	NA	Diameter sum per species/ Total diameter sum for plot	Sum of midpoint dimeter per species, divided by sum for all species	Sum of DBH per species divided by total sum in the plot	Sum of DBH per species divided by total sum in the plot
Saplings	Data	NA	NA	0.1 Ha circular plot	NA	NA	NA
	Manimu lation	NA	NA	Sapling /adult ration for each specie	NA	NA	NA

Table D. Summary statistics for topographic and climatic variables used. Statistics are estimated for the extent of the foothill zone only.

	Mean	Std Dev	Min	Max	Quantile 25	Quantile 75
Elevation (ma.s.l)	1344.15	357.35	431.50	1862.00	1078.18	1647.39
Slope (%)	27.79	8.63	0.00	61.06	23.15	33.25
Aspect (degrees)	South (194.44)	71.52	-0.18	346.24	142.86	250.48
Distance to road (km)	2.592	3.674	0.00	20.287	0.598	2.838
Distance to Stream (m)	227.31	184.18	0.00	1228.15	90.88	315.96
Temperature during the growing season (°C)	6.64	2.29	0.17	11.08	4.76	8.51
Temperature during non-growing season months (°C)	18.01	2.66	10.68	23.39	15.90	20.16
Precipitation during the growing season (mm)	131.43	15.87	92.45	166.63	119.63	145.53
Precipitation during non-growing season months (mm)	21.64	2.40	14.67	27.38	19.83	23.43

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