

Energy Research and Development Division
FINAL PROJECT REPORT

**PROJECTED CLIMATE CHANGE
EFFECTS ON VEGETATION AND
HYDROLOGY IN
CALIFORNIA CHAPARRAL AND
SIERRAN CONIFER ECOSYSTEMS**

Implications for Hydroelectric Generation

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PREFACE

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ABSTRACT

Hydroelectricity generation is affected by the interactions of fire, vegetation, climate, and hydrology. Fire frequency and intensity in California is changing, and these changing conditions could result in the conversion of chaparral shrublands to exotic annual grasslands, and at higher elevations from conifer forests to shrublands. This study investigated the relationship of fire to vegetation change in southern California chaparral and Sierra Nevada conifer forest and potential consequences of vegetation change for watershed hydrology. It simultaneously evaluated ecophysiological responses of dominant plant species to seasonal climate and drought in both ecosystems. Species level information was used to parameterize a coupled model of ecosystem carbon cycling, vegetation growth, and spatially distributed hydrology for predicting watershed yield under different vegetation. The study also found that short interval fire, while increasing in occurrence in southern California, is not leading to widespread vegetation change. In the Sierra Nevada, researchers detected an important role for post-fire climate, aspect, and elevation in governing rates of forest regrowth. In studying the consequences of species change for water yield, they compared water use and carbon gain by dominant chaparral and mixed conifer species. In chaparral, they found that species varied in their water use and response to seasonal drought but that these differences did not translate to watershed differences in stream flow. However modeling suggested that if chaparral watersheds were converted entirely to grasses, water yield would rise dramatically. In the Sierra Nevada ecophysiological studies demonstrated that shrubs were less conservative in their water use than trees. The watershed model suggested that water yield would decrease slightly if trees were eliminated. Conversion from trees to shrubs would likely produce a moderate increase in streamflow, however, downstream reservoir storage would only increase during dry years. Additional hydropower generation would be limited to winter months.

Keywords: Eco-hydrology; Drought stress; Plant water use; Vegetation type conversion; Reservoir modeling; Post-fire vegetation recovery.

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EXECUTIVE SUMMARY

Introduction

Hydroelectric power is an important contributor to California's clean energy strategies. The amount of electricity generated by hydrodams varies from year to year with changes in weather and the resulting runoff. It is affected by the complex interactions of wildfire, vegetation, climate, and hydrology and all are projected to change in the future. Change in vegetation types is driven by periodic disturbance, such as fire, and by longer-term effects of climate change. This can dramatically alter the timing and volume of streamflow in watersheds that supply California with much of its water. Gaining a better understanding of how trees, shrubs, and grasses use water and tolerate drought stress today is essential to make accurate predictions of water availability for hydropower and other uses.

Most previous studies have investigated the potential impacts of climate change on vegetation, wildfire, or hydrology separately. This study investigated the relationship of fire to vegetation change in southern California chaparral shrublands and in Sierra Nevada conifer forest and considered climate factors that might influence response to fire. It simultaneously evaluated how trees, shrubs, and grasses use water throughout the year and during drought in both chaparral and Sierra Nevada ecosystems. The study predicted how changes in vegetation after wildfire would affect future streamflows and the capacity of a hydroelectric reservoir.

Project Purpose

Relatively little is known about the extent that interactions between wildfire, vegetation recovery, and climate affect water runoff from California ecosystems and how that might impact hydroelectricity generation. This project was designed to inform managers of hydroelectric facilities and energy forecasters how these likely changes might affect hydropower resources. The study:

- Determined the role of fire and climate in influencing vegetation change in southern California chaparral and in montane mixed conifer forests of the Sierra Nevada.
- Evaluated ecophysiological responses of major plant species in chaparral and mixed conifer forests to seasonal drought and soil water availability.
- Projected possible changes in watershed streamflow as a result of changes in the vegetation of chaparral and mixed conifer ecosystems.
- Analyzed potential changes in hydroelectric reservoir storage while still meeting instream flow needs for other uses.

Project Process and Results

Fire and climate change affects vegetation

The researchers examined areas previously burned to determine if frequent fire causes long-term changes in vegetation in southern California chaparral and Sierra Nevada conifer forest. Chaparral recovery or loss, following fire was studied in southern California using two methods: (1) high resolution, historic aerial photos, and (2) a series of satellite image data during several years. Based on evidence in the literature, the researchers expected that repeated

fires within several years (termed 'short interval fire'), would impede chaparral recovery. They specifically tested whether short interval fire could lead to converting chaparral to exotic annual grassland. They selected southern California chaparral, particularly in the region from Ventura to San Diego to evaluate this hypothesis. These areas are subjected to Santa Ana winds that drive wildfire spread. If type conversion is going to occur, this is an area where it could be expected.

The high-resolution aerial photography study was conducted in Ventura and Los Angeles counties. The authors found that approximately 10 percent of the region has experienced a short interval fire (2-5 years between fire events) during the past 90 years and that although the number of such fires fluctuated over the last eight decades, the area burning in such fires has consistently increased. Nonetheless analysis of vegetation change in sites experiencing short interval fire versus once burned sites did not reveal significant loss of chaparral. In fact, chaparral shrublands appear to be maintaining their dominance within the sites examined. However, invasive grasses have increased in all burned areas, particularly on south facing slopes. It was concluded that a single short interval fire was not adequate to cause significant chaparral loss.

Using a series of Landsat Thematic Mapper satellite imagery from 1985 to 2009 for a much broader area of southern California, vegetation recovery was compared between areas burned twice in eight years and areas burned only once. Correlations between recovery and fire history, climate, and elevation were also examined. Reduced chaparral recovery was found in two lower elevation areas that were burned twice in eight years. However, extensive conversion of chaparral to grassland was not evident. In most cases vegetation cover was similar between once burned and twice burned areas, reinforcing the findings from the aerial photography analysis.

Forest cover in the Sierra Nevada experiences more severe and more persistent changes following wildfire, providing lasting impacts on hydrology. A series of Landsat image data from 1994 to 2011 for the Sierra Nevada was used to monitor post-fire vegetation recovery in mixed conifer and red fir forests. Fire severity, post-fire climate variables, and topography were used to model changes in vegetation in the five years following 35 wildfires. Lower elevation mixed conifer forests showed faster recovery in vegetation, but were sensitive to post-fire drought. Vegetation recovery in red fir forests found at higher elevation was more sensitive to extreme high temperatures after fire. Climate appears to play a critical role in short-term vegetation recovery following fire, and changes in climate may prolong or worsen vegetation recovery.

Vegetation change and drought affect how much water plants need

In the Santa Barbara foothills, researchers measured changes in water availability and use by important chaparral species during the wet and dry season. Since both study years were during exceptional drought, researchers had a rare opportunity to assess the degree to which these 'drought tolerant' shrub species actually responded to drought conditions. The shrub species in the chaparral completely stopped transpiring (emitting water from the leaves) when, near the end of each dry season, available water was limited. Photosynthesis, however, resumed

following both brief, but intense, winter rainfall events as well as summer fog events, which are known to relieve water stress. These observations suggest that the shrub species in this study can acquire water from relatively shallow levels in the soil profile. Species varied in their responses to drought. One species, chamise, was noticeably more drought tolerant than the ceanothus or manzanita species studied. In sum, field-based observations of seasonal changes in plant physiological activity highlights that water is a prominent limiting resource to ecosystem productivity in this California region.

During the 2014 growing season, researchers measured ecophysiological responses (plant water status and leaf gas exchange rates) of two major tree and shrub species to changes in seasonal water availability. Measurements were made at two sites within the southern Sierra Nevada Critical Zone Observatory located on the North Fork of the Kings River not far from a hydroelectric facility at Shaver Lake. Plant physiological observations were used to model the impact of changes in seasonal water availability and vegetation type-conversion on streamflow. Based on field observations, transpiration rates of shrubs was greater than that of trees through the peak growing season. However, neither shrubs nor trees completely stopped transpiring at any point in the season, suggesting that these plants could continue growing even as residual soil moisture declined.

Vegetation change and climate warming affects streamflow and reservoir capacity

Researchers used a model that coupled ecosystem carbon cycling, vegetation growth, and hydrologic processes to estimate how vegetation change in southern California and Sierra Nevada might impact hydrology. They incorporated analysis from the field monitoring of forest and shrub water use into the model and then used the model to estimate differences between species in their water use in case study watersheds. For the Sierra Nevada, watershed conversion from forest to shrub resulted in a modest increase in streamflow (average of 19 percent during historic climate variation). They note, however, that this is likely a maximum potential change since it reflects complete conversion of the whole watershed from forest to shrub. For the southern California chaparral, 100 percent conversion to grass leads to a substantial increase in flow (average of 85 percent over historic climate variation). The authors also examined climate-warming scenarios and found, similar to previous studies that a 3°Celsius (5.4° Fahrenheit) temperature increase could lead to a shift in the timing of streamflow for the Sierra site but had little impact for the Southern California site. In both watersheds, vegetation conversion could cause greater changes in total annual streamflow than changes due to a moderate climate warming.

For the Sierra site the researchers also linked changes in streamflow with a reservoir model to demonstrate potential implications for meeting reservoir management objectives. Three change scenarios were used to investigate the effect of 1) tree-to-shrub type conversion, 2) a 3°C temperature increase and 3) both type conversion and a temperature increase combined. A reservoir simulation model was used to test the effects of three change scenarios on hydroelectric capacity. Following conversion from trees to shrubs (scenario 1), annual maximum storage increases in the reservoir. However, the increase is generally minimal during wet years because there is little excess winter storage capacity for the extra streamflow

generated following type conversion. In dry years, storage is available to capture the additional inflow. Climate change in the absence of vegetation change (scenario 2) produces a reduction in annual maximum storage because more water moves through the reservoir during the winter when storage is limited. For scenario 3, the increases in annual maximum storage that were produced following conversion to shrubland were reduced compared to scenario 1, as the 3°C increase in temperature caused an earlier shift in streamflow timing and partially canceled out the effect of increased storage with shrubland. These modeling results suggest that unless there is sufficient reservoir capacity to capture additional winter flows generated by type conversion, increases in hydropower generation will likely be limited to the winter months when demand for hydropower is often lowest.

Project Benefits

Understanding these complex issues will improve the ability of water managers and energy planners to predict the effects of wildfire on vegetation conversion and thus on the availability of water for hydropower operations. These issues will become more critical as climate change affects water supplies and the frequency and severity of wildfire and vegetation recovery. Linking strategic field data collection and models can be used as an effective tool for assessing the potential impact of vegetation change on the water balance of an ecosystem. This is an increasingly valuable approach to inform management decisions focused on adaptation strategies based on projected changes in climate and can provide more accurate baseload hydroelectric generation forecasts.

CHAPTER 1:

Introduction

Past work has shown that climate change is likely to cause more frequent and/or larger fires in many parts of California, by creating more severe fire seasons or increased fuels in sparsely vegetated areas (e.g., Lenihan et al. 2006, Westerling et al. 2009, Krawchuk and Moritz 2012). Adapting to future fire regimes is therefore one of the key challenges that California faces (Moritz and Stephens 2006). This will be made even more difficult by interactions between fire and invasive species (D'Antonio and Vitousek 1992, Brooks et al. 2004), which can be exacerbated by the increasing atmospheric nitrogen deposition that has been documented throughout many parts of the state (Weiss 2006). Increasing ignitions with further development (Syphard et al. 2007) will also continue to modify natural fire regimes across California. As climate and fire regimes change, vegetation patterns, habitat quality, and many related ecosystem services will also be impacted in complex ways.

Across vast chaparral-dominated portions of southern and central California, the most likely vegetation type conversion of concern is related to increasing fire frequencies and expanding invasive annual grasslands (e.g., Zedler et al. 1983). In many montane regions of California, although not necessarily driven by non-native species, future shifts from forests to shrublands may be more likely, which can then be maintained by pyrogenic feedbacks after such vegetation transitions have occurred (e.g., Odion et al. 2010). Despite clear links to hydrologic cycling (e.g., water use by plants, snowpack dynamics, and stream water quality and quantity), relatively little is known about how vegetation type conversions in different environments may alter downstream flows. Fine-scale field studies are one avenue for filling this knowledge gap at localized sites; however, watershed-scale modeling is needed to link fire, vegetation, and ecohydrology across larger areas.

The application of new process-based hydro-ecological models (e.g., Tague et al. 2009) provides a tool for examining how habitat alteration and vegetation type conversion in different regions of California may affect the timing, amount, and quality of streamflows and resulting hydropower potential at the watershed scale. The work documented here uses the Regional Hydrologic Ecosystem Simulation System (RHESSys) model (see <http://fiesta.bren.ucsb.edu/~rhessys/>), to simulate vegetation type conversion scenarios in key case study areas of southern coastal California (currently shrubland dominated) and the Sierra Nevada Mountains (currently forest-dominated). Hydro-ecological modeling parameters were informed by extensive field-based measurements of actual water availability to, and water use by, dominant plant species typical of these environments. In addition, regional scale analyses of past vegetation type conversions and potential causal drivers were also performed to assess the degree to which such shifts may already have occurred.

Besides the inherent ecological interest in these topics, this study was motivated by the direct application of this new information to management of hydropower dams for generating electricity balanced with other competing uses of water. Consequently the study also analyzed

how conversion from forest to shrubland, climate warming, or both, would affect reservoir storage while meeting instream flow requirements for other uses.

CHAPTER 2: Chapparral Ecosystem Response to Fire and Drought

2.1 Rationale

Shrubland ecosystems in southern California have been increasingly reduced in area due to the expansion of urban and suburban development over the last several decades. Simultaneous with this growth, fire frequencies appear to be increasing (Keeley and Fotheringham 2001, Syphard et al. 2007) and the region is experiencing severe drought (www.droughtmonitor.unl.edu). The primary natural disturbance regime in these shrublands is one of periodic stand-replacing crown fires that occur with 20-50 year return times depending on the specific area (Keeley and Fotheringham 2001). During periods between fire, a diverse community of native shrubs develops a closed canopy with little understory. At maturity these shrubs are 1.5 to 3 meters tall with dense intertwined canopies. Recent predictions of climate change in California suggest that fire frequencies will increase as climate change continues (Moritz et al. 2006). The occurrence of severe drought, both after fire, and in mature stands, could influence species composition by causing selective mortality of particular species opening up the plant canopy. Likewise, shorter return interval fires may lead to loss of shrub species and to more open grassland or savanna like vegetation. Thus altered fire regimes and drought could lead to local scale species loss and vegetation change. The consequences of such changes for landscape scale carbon sequestration and water use are poorly understood.

Chaparral is a community of woody, usually sclerophyllous-leaved shrubs that dominate cismontane reaches of California's Pacific coast, as well as the foothills of the Sierra Nevada. These regions experience a cool season rainfall regime where rain occurs between late October and early May and summers are hot and dry. Fires can occur year round in these shrublands but the largest fires typically burn between July and late October (Keeley and Fotheringham 2001). A diversity of shrubs has developed physiological adaptations to tolerate seasonal drought and persist with current fire regimes. The physiological mechanisms through which different species respond to extreme climate events such as strong seasonal or extreme duration droughts or to fire will ultimately determine population persistence as climate and fire regime change.

Chaparral shrub species are typically considered to be drought-adapted sclerophylls, and it has been demonstrated that their photosynthesis and stomatal conductance behavior respond strongly to water stress (e.g., Kolb & Davis 1994, Jacobsen et al. 2007 a, b). Yet few studies have evaluated photosynthesis, stomatal behavior, leaf fluorescence, and water stress across severe drought years to evaluate how these basic physiological functions change and at what point during extreme droughts shrubs cease to photosynthesize and thus potentially become carbon limited. Recent climate analyses suggest droughts are becoming more severe and prolonged in parts of the western USA (Weiss et al. 2009, Cook et al. 2004) and understanding species level variation in how shrubs will respond to these severe droughts will help to predict future compositional changes. Shrub mortality has been documented after drought for a few chaparral species (e.g., Pratt et al. 2014). While it is generally agreed that hydrologic failure is the main

cause of woody plant death (Jacobsen et al. 2007a,b), low rates of photosynthesis could contribute to poor osmoregulation and slow carbon starvation as a drought become lengthy or severe. Thus understanding physiological responses of different species to extreme drought could help to understand variation in species mortality patterns.

Another potential cause for species loss within chaparral and potentially chaparral conversion to grassland is shortened fire return intervals. Chaparral species can be broadly categorized as being either obligate seeders, which cannot resprout after fire, and post-fire resprouters. Resprouters depend on a carbon-rich burl or lignotuber, from which they resprout after fire. Obligate seeders do not resprout, but rely on seeds which are stored in the seed bank and require fire disturbance for germination. Fire tends to stimulate release of these dormant seeds depleting the stored seedbank. This seedbank is restored slowly over the subsequent decades because shrubs typically take four or more years to become reproductive (Keeley 1986) and a large proportion of the annual seed production is removed by animals (Zammit and Zedler 1988, Mills and Kummerow, 1989). Thus, as fire return frequencies shorten, obligate seeders are at risk of seed bank depletion if shrubs cannot reach sexual maturity and set seed before another fire occurs (Zedler 1995). By contrast, non-native annual grass species can recolonize quickly after fire even though their seeds can be killed by high fire temperatures. The prevalence of invasive annual grasses across southern California landscapes, particularly within disturbed shrublands, thus provides a source of rapid fire ignition (Syphard et al. 2007), quickly carrying fire into adjacent shrublands. Invasive annual grasses can reduce the growth of germinating chaparral species (Beyers 2004), effectively lowering their recruitment and frequency across the landscape. It has thus been hypothesized that short return interval fires may eliminate chaparral species from the landscape, effectively resulting in conversion of chaparral communities to savanna or grassland habitat.

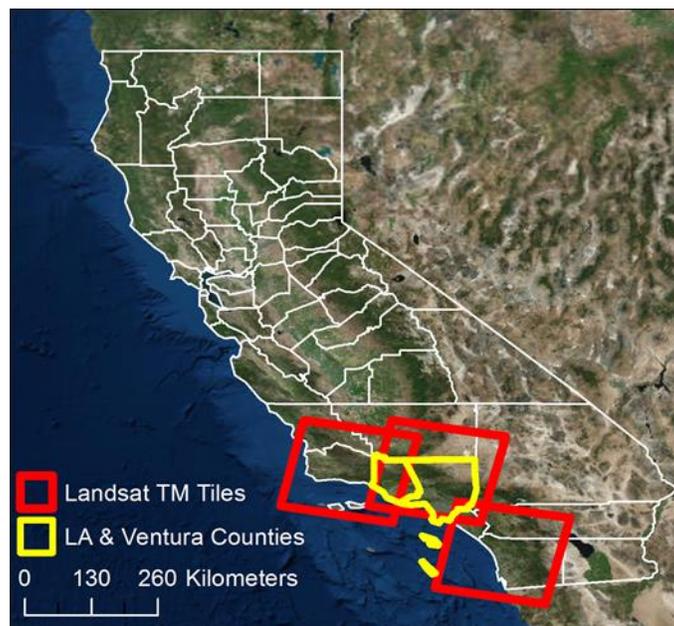
2.2 Designation and Description of Study Region

Chaparral shrubland, a vegetation type dominated by evergreen, tough leaved shrubs, occurs on mountain slopes from northern Baja California to south central Oregon (Keeley and Davis 2007). It generally occurs above the lower elevation sage scrub habitats that typically extend from sea level to 300 m but below conifer forests (>1500 m). It reaches its greatest development in the Transverse and Peninsular ranges of central California and in the foothills of the Sierra Nevada immediately downslope of the mixed conifer belt. It ranges over a wide latitudinal belt of approximately 10 degrees and with that spans a climate gradient from more xeric slopes in San Diego County (250-400 mm MAP), to mesic rocky slopes in Mendocino County with greater than 2000 mm of annual precipitation. This study focuses on the more xeric portions of this gradient.

Chaparral habitat in southern California tends to occur on rocky slopes with poor soil development and relatively limited water storage (Keeley and Davis 2007). The more arid nature of the habitats supporting chaparral in southern California, and the high human population in the region make shrublands there more likely to be susceptible to altered fire regimes (Keeley 2002). For this reason, the researchers selected south central and southern California shrublands for this study of how altered fire regimes may influence vegetation

conversion in shrubland communities. The researchers examined potential conversion of chaparral to grassland at two scales. High spatial resolution aerial imagery can identify change at shrub-to-patch scales. Aerial image analysis described in sections 2.3 and 2.4 focuses on Ventura and Los Angeles (LA) Counties where Santa Ana winds drive wildfire and where ignition rates are increasing (Keeley and Fotheringham 2000) (Figure 1, areas delineated in yellow). Due to limited availability of historical aerial imagery over the wider extend of southern California and the need to understand landscape-scale changes in chaparral cover, the researchers also used Landsat Thematic Mapper (TM) data covering southern San Luis Obispo County, all of Santa Barbara, Ventura, LA and Orange counties, the western half of Riverside County and most of San Diego County (Figure 1, area delineated in red). Despite the wide coverage across the region, most short interval fires were located in LA, Orange, and San Diego Counties.

Figure 1: Map of Areas in California Where the Effects of Short Interval Fire on Vegetation Were Examined



2.3 Fire overlap map & occurrence of short interval fires

To determine the occurrence of short interval fires in Southern California, the researchers mapped the minimum fire interval (i.e., shortest duration between two fires) within the study site of Ventura and Los Angeles counties. They first acquired fire history data (1878-2009) from the Fire and Resources Assessment Program (FRAP) database (CALFIRE, www.fire.ca.gov). Then they clipped the fire history shapefile to the study site boundary to select for fires that fell within Ventura or Los Angeles County. With these selected fires, all fire perimeters were overlapped and the shapefile was merged using the “spaghetti and meatballs” technique (Honeycutt 2012) in ArcMap (ArcGIS 10.1). This enabled them to create a new shapefile of the merged fire perimeters with unique fire histories while preserving all original fire perimeter

information, such as “Fire Year” and “Alarm Date”. The attribute table was exported to Microsoft Excel (Microsoft) and metrics such as “Minimum Fire Interval” and “Number of Fires” were calculated. Unique fire perimeters were edited if they had two “Alarm Dates” in the same year to eliminate errors due to a single fire being reported by multiple agencies. This modified table was then joined back to the merged shapefile in ArcMap. The resulting polygons in this merged shapefile reflected the areas that had overlapping fires derived from the original FRAP fire perimeters and contained the complete fire history information for all fires that had occurred at that location.

To investigate the spatial extent of short interval fires within the chaparral study site, the authors grouped all polygons by their shortest fire interval: 2-5 years, 6-10 years, 11-20 years or >20 years (Figure 2, Table 1). Urban and agriculture areas were omitted.

Figure 2: Minimum Fire Intervals within Ventura and Los Angeles County.

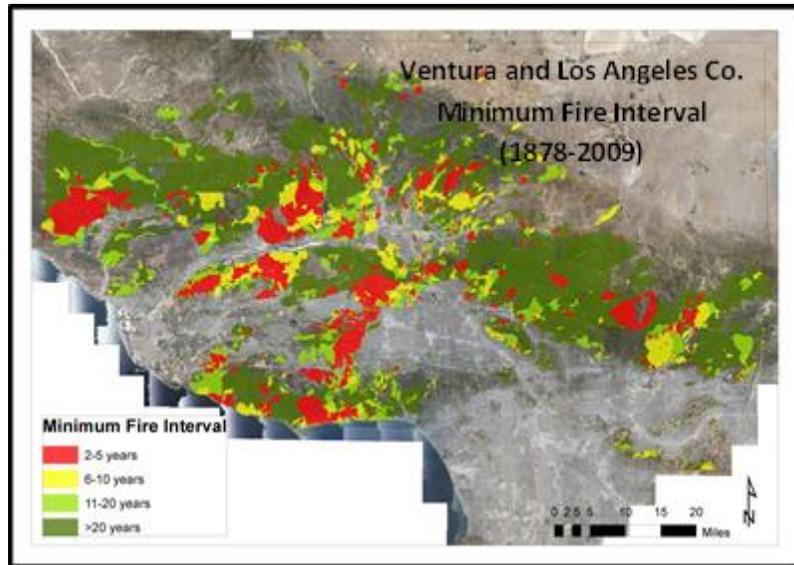
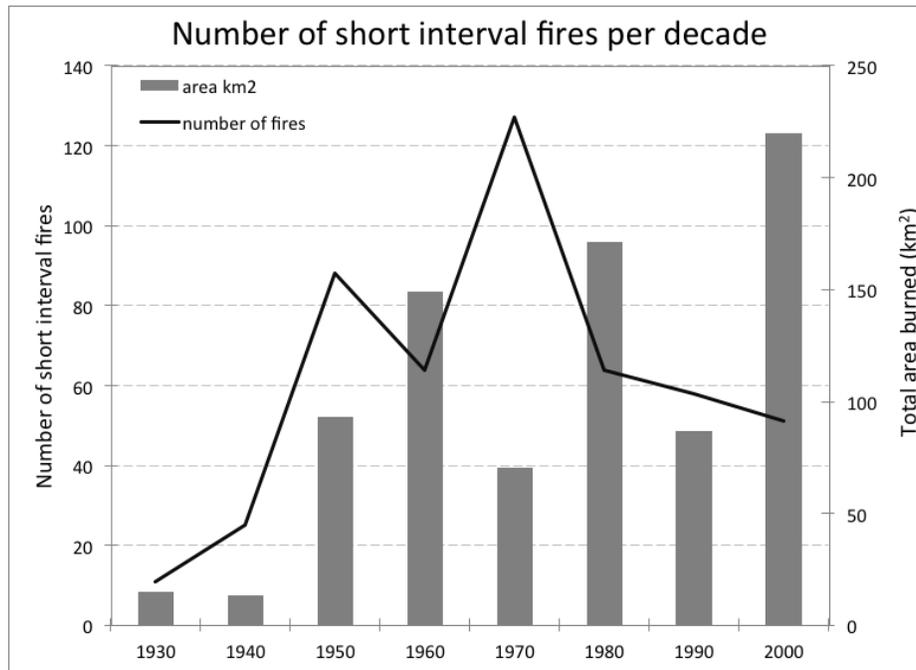


Table 1: Total Area of Minimum Fire Intervals within Ventura (5,720 km²) and Los Angeles (12,310 km²) County

MINIMUM FIRE INTERVAL	AREA (km ²)	AREA (%)
2-5 years	687.31	10.16
6-10 years	416.31	6.15
11-20 years	519.09	7.67
>20 years	5,142.69	76.01
total area burned at least once	6,765.40	100.00

To examine the temporal aspect of short interval fires within the study site, they tallied all “Alarm Dates” by decade (Figure 3). Area burned was determined for each polygon in ArcMap, exported, and total areas calculated in Microsoft Excel.

Figure 3: Number of Short Interval Fires by Decade within Ventura and Los Angeles County



*Bin "2000" includes years 2000-2009 making 2005 the last year to be included in the "two fires within five years" interval. Fires recorded after 2005 have a fire interval of less than five years.

2.4 Effects of short interval fire using aerial photographs

2.4.1 Selection of study region and overlap pairs

To quantify the effect of a short interval fire on shrublands in Ventura and Los Angeles counties, adjacent paired polygons ("sites") that experienced either one fire or two fires within the same five-year period were identified. Polygons that experienced two fires within five years were classified as "short interval fire" samples. Polygons that experienced one fire within the same five-year period were classified as "historic interval fire" samples. These polygons had, on average, experienced a fire 28.6 years prior to this study and two of the eleven sites had no prior record of fire. Additionally, all polygons had to be larger than 0.5 km² (50 hectares) and were selected throughout the two counties to include inland conditions (more arid) as well as coastal conditions (more mesic) as well as within the Santa Ana corridor (extreme wind events) (Figure 4).

Polygons that experienced one fire within five years were labeled "once burn" and polygons that experienced two fires were labeled "twice burn". Twice burn polygons experienced the first and the second fire, whereas the once burn polygons only experienced the second, more recent, fire. Once burn polygons were selected from the second fire to capture the same number of regrowth years as the twice burn polygons. If there was no adjacent once burn polygon that burned in the second fire, a once burn polygon could have been selected from the first fire year only if ≥ 19 years of regrowth had occurred. This exception was allowed assuming that any

difference in vegetation cover would be negligible after ≥ 19 total years of regrowth. Table 2 includes fire information for the eleven sites.

Figure 4: Study Sites within Ventura and Los Angeles County. The Eleven Sites are Delineated with a Black Line Along the Margin of the Polygon

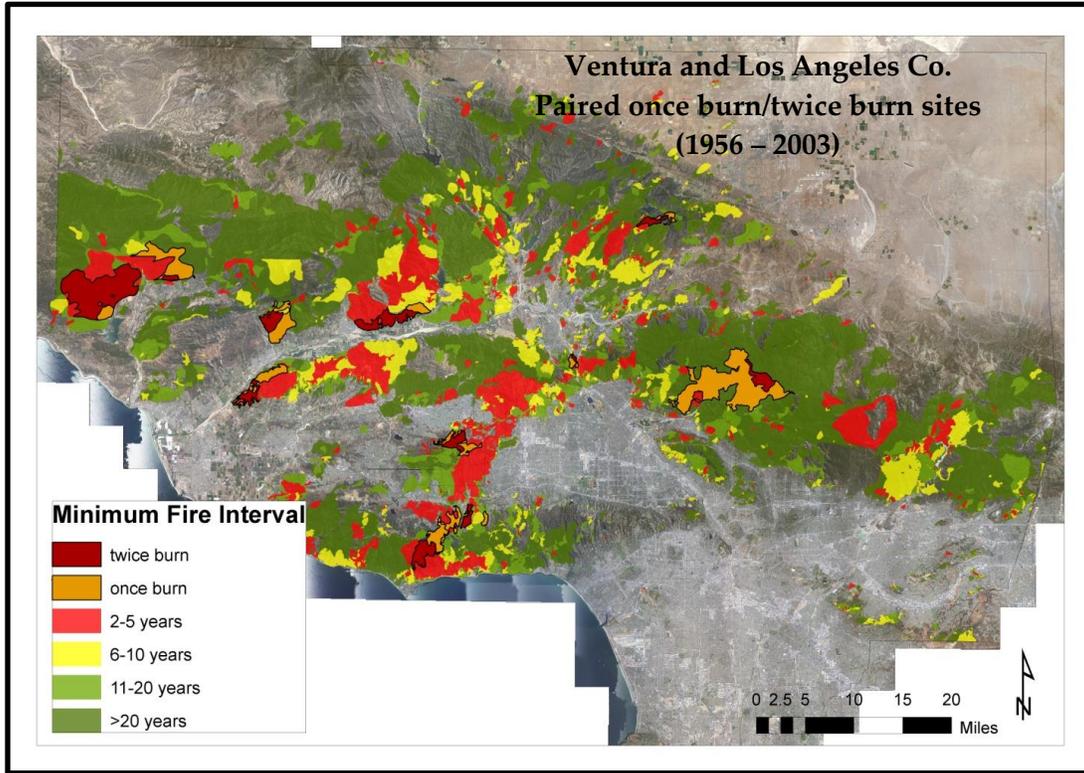


Table 2: Details of the Study Sites within Ventura and Los Angeles County

	SITE ID	PRE FIRE IMAGE	MINIMUM FIRE INTERVAL	POST FIRE IMAGE	YEARS OF REGROWTH	AVG. DISTANCE FROM COAST (km)
1	001	1961	1962, 1967	1975	8	26.08
2	002	1994	1998, 2003	2009	6	42.29
3	004	1961	1962, 1967	1980	13	19.57
4	006	1964	1985, 1990	2009	19	21.01
5	103	1956	1975, 1979	2002	23	45.51
6	104	1959	1960, 1964	2002	38	60.72
7	109	1959	1967, 1970	1980	10	35.50
8	110	1964	1967, 1970	1977	7	21.29
9	111	1976	1980, 1982	1994	12	8.60
10	112	1978	1985, 1990	2009	19	10.93
11	113	1952	1956, 1958	1980	22	5.58

2.4.2 Acquisition of aerial photographs & georectification

Historical aerial photographs were acquired from the Map and Imagery Laboratory at University of California Santa Barbara (www.library.ucsb.edu/mil). The objective of using aerial images was to quantify the amount of change in vegetation cover between sites that experienced one fire and sites that experienced two fires within the same five year period.

Prefire images were ideally taken one year before the first fire to reflect vegetation cover present at the time of the first fire. Postfire images were taken six years or more following the second fire to capture maximum vegetation regrowth cover without encountering a third fire. Final image selection was determined by the availability of images for the location and timeframe of interest. The temporal extent of fire events spanned from 1956 to 2003 and the corresponding aerial images were captured between 1952 and 2009.

To quantify prefire vegetation cover and compare it to postfire vegetation cover on a pixel-by-pixel basis, we georectified all aerial images to the same base image in ENVI (Exelis Visual Information Solutions). We used a 2009 grayscale, 1-meter spatial resolution, digital orthophoto quarter quad (DOQQ) of Ventura or Los Angeles County, collected by the United States Geological Survey, as the base image. Each acquired aerial image was warped using ground control points (GCPs). Temporally stable objects such as shrubs, trees, rocky outcrops, and crests and troughs of the mountainous landscape were used as reference points, observed at 12 times their original size. Dirt roads and permanent structures were also used, although these more permanent features were rare in the aerial images. The terrain of the aerial images was mountainous and highly variable so GCPs were placed at a high density to increase warping accuracy: 300 – 600 GCPs per aerial image was common. Each aerial image was warped to triangulation, resampled to the nearest neighbor, and resulted in an output georeferenced image with 1 meter spatial resolution.

Warped georeferenced images were then hand selected and mosaicked together to minimize edge distortion and to ensure the most accurate image for further vegetation analysis. All eleven sites were warped and mosaicked to capture the entire prefire and postfire once burn and twice burn polygons. Only two sites (006 and 103) were not georeferenced in full due to a lack of available images and/or due to their extensive size. For site 006, the entire twice burn polygon was georeferenced and an equivalent area of the once burn polygon was also georeferenced. For site 103, the entire twice burn polygon was georeferenced and approximately four times its area in the once burn polygon was georeferenced. Finally, mosaicked images were validated for their spatial accuracy using the 2009 DOQQ as the base map. Forty to 100 GCPs were identified per site with a root mean square error of <10 pixels (i.e., < 10 meters).

2.4.3 Change detection techniques

Once prefire and postfire images were mosaicked and validated, random points were generated in the mosaicked prefire image to select subsites to analyze for vegetation cover. Each subsite was 50 x 50 pixels (50 x 50 meters) and had either a north or south-facing aspect (north: 0.0° to 67.5° or 292.5° to 360°; south: 112.5° to 247.5°). Approximately sixteen subsites were randomly selected per site (once burn polygon: 4 north, 4 south; twice burn polygon: 4 north, 4 south).

These subsites were then adjusted to fit entirely on one aspect, ensuring they did not overlap a mountain ridge or valley. Aspect was verified with 30 meter spatial resolution USGS Digital Elevation Model (DEM) data and/or visually with Google Earth. All prefire subsites were replicated in the mosaicked postfire image to capture vegetation cover at the same location.

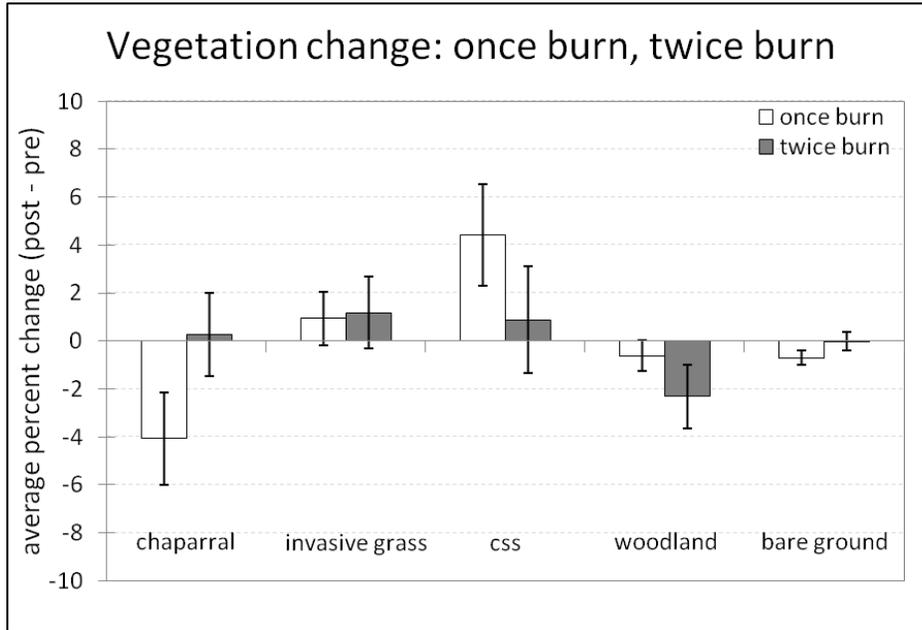
To quantify vegetation cover at each subsite, the “dot grid” method was used (Floyd 1982; Dublin 1991). A 10 x 10 grid (100 points) was overlaid on each subsite with a spacing of 5 pixels (5 meters) between each data point. Vegetation cover was recorded at each data point while being observed at 5 times its original size. Vegetation cover was classified to life form: chaparral, invasive grass, coastal sage scrub, woodland, or bare ground/exposed rock. To improve classification accuracy, researchers took into consideration the solar zenith angle to account for shadows and referenced Google Earth for current cover and seasonal changes. They also performed verification field trips. They then tallied the total number of data points within each classification to quantify percent vegetation cover. Prefire vegetation cover values were subtracted from postfire vegetation cover values to quantify percent vegetation change. By comparing percent vegetation change in once burn and twice burn sites, the researchers were able to quantify the impact of two fires in five years compared to one fire in five years. Significance was determined by performing a Wilcoxon Signed Rank test on each vegetation class in JMP software.

2.4.4 Results

Overall, there is no strong indication that vegetation cover is different after one fire or two fires within the same five year period in Southern California from 1956 to 2003 (Figure 5). All vegetation classes show no significant difference in percent change of vegetation cover when comparing areas that experienced one fire and areas that experienced two fires.

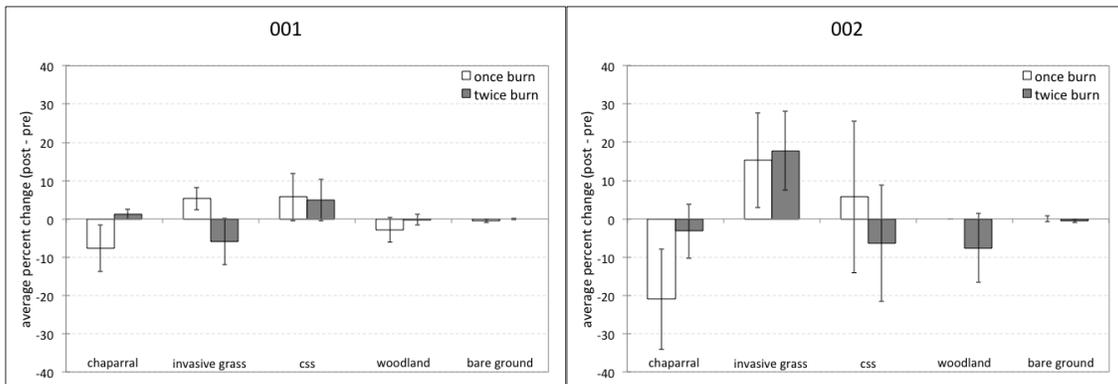
The results show chaparral slightly decreases following one fire, while there is almost no difference between prefire and postfire percent covers following two fires. Invasive grass cover does not change or slightly increases in cover following any amount of fire. Coastal sage scrub cover shows a slight increase in cover following one fire, while there is no difference between prefire and postfire percent covers following two fires. Woodland cover slightly decreases or has no change in cover following one fire and slightly decreases following two fires. Bare ground and exposed soil slightly decrease following one fire and does not change following two fires.

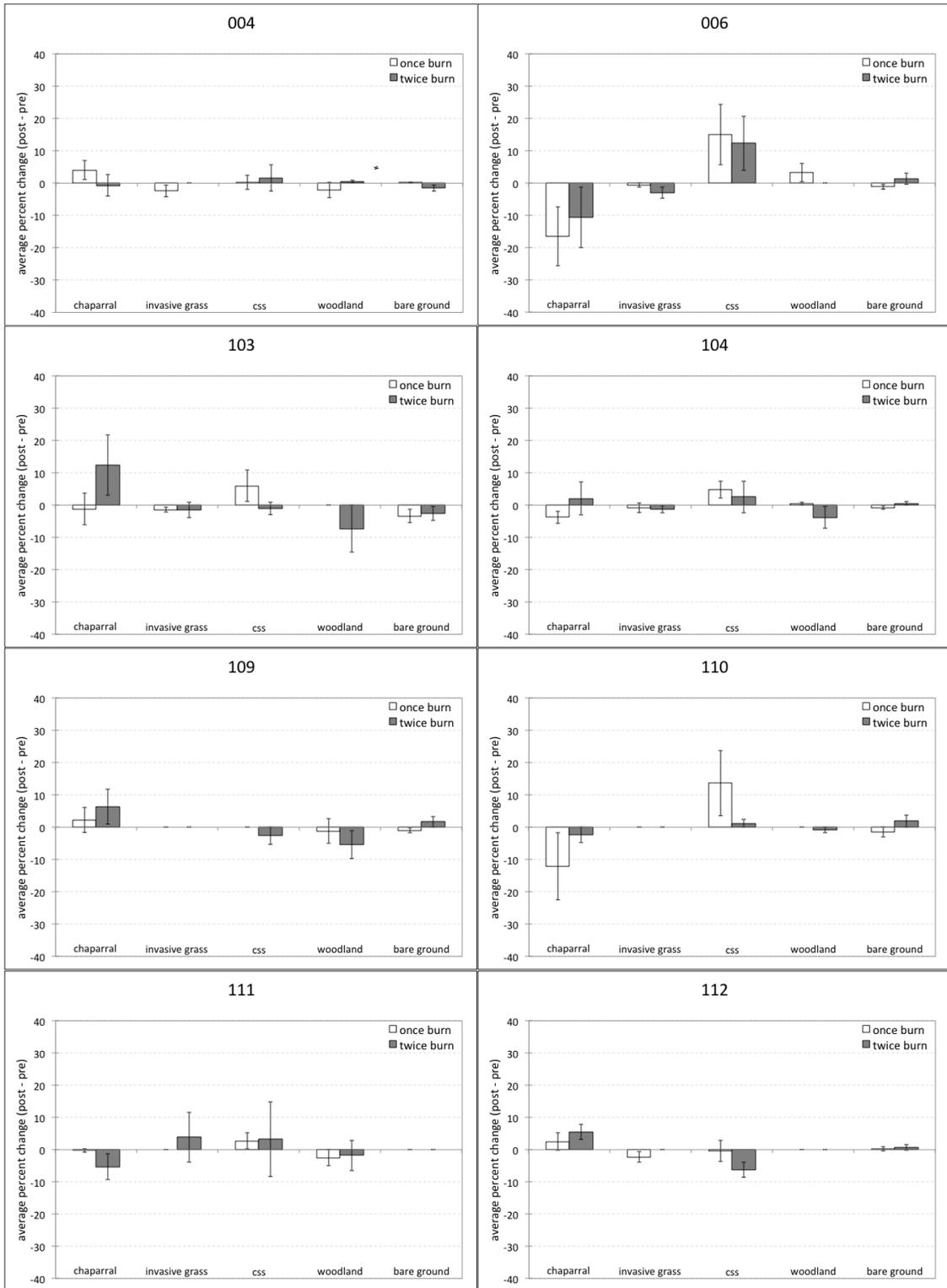
Figure 5: Average Vegetation Change (%) Following One or Two Fires in the Same Five Year Period

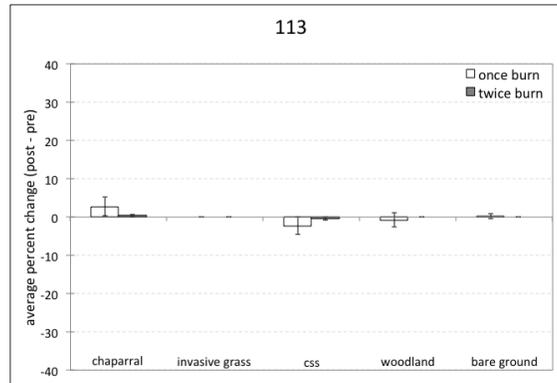


To further investigate the difference between one fire and two fires within five years, the authors analyzed the average percent vegetation change at each of the eleven sites. The results show a large amount of variation in response to fire with no significant differences between once burn and twice burn polygons, with the exception of bare ground at site 104 (Figure 6).

Figure 6: Average Vegetation Change (%) Per Site Following One or Two Fires Within the Same Five Year Period

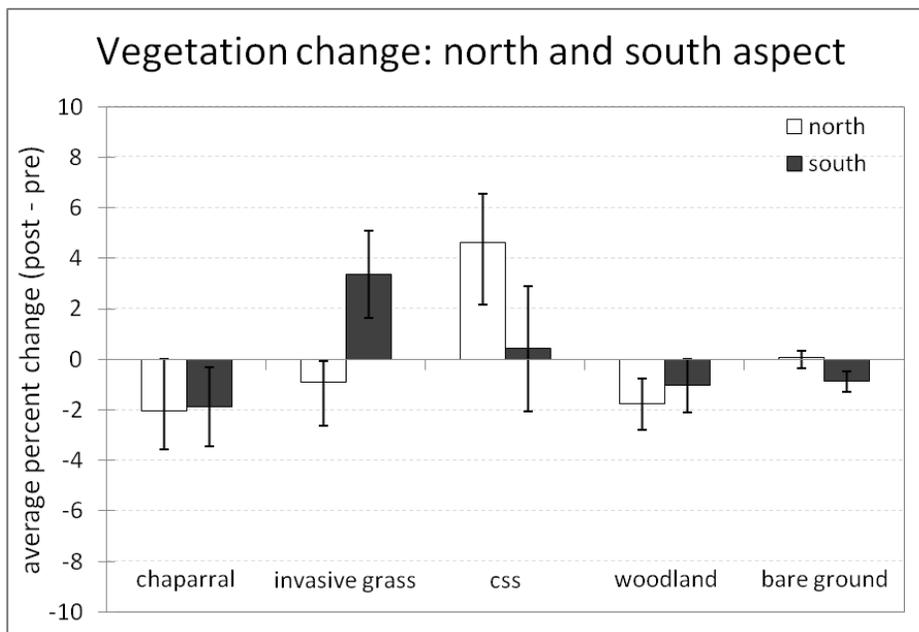






Researchers also examined percent change in vegetation cover by aspect (Figure 7). Invasive grass increases significantly on south-facing aspects ($p < 0.02$) compared to north-facing aspects. Coastal sage scrub cover increases on north-facing aspects and does not change on south-facing aspects. Chaparral and woodland cover show no change to a slight decrease on both north and south-facing aspects. Bare ground and exposed soil is not affected by aspect.

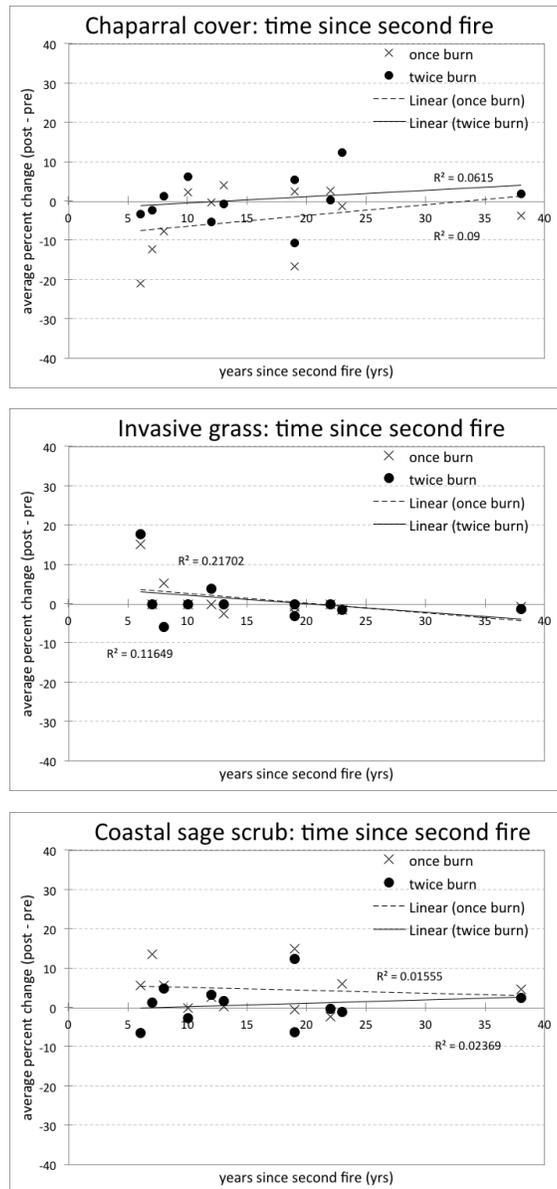
Figure 7: Average Vegetation Change (%) on Northern or Southern Facing Aspects



To investigate if the observed percent change in vegetation cover is correlated with the “time since fire” they plotted the years of regrowth (number of years between the second fire and postfire image) by the average percent change in vegetation cover (Figure 8). They focused on chaparral, invasive grass, and coastal sage scrub since those are the main vegetation communities of interest in Southern California. While the number of regrowth years is not a strong explanatory variable for the percent vegetation change observed, there are trends among the data. The percent change in chaparral cover tends to increase over time, the percent change

in invasive grass cover generally decreases over time, and the percent change in coastal sage scrub cover has a mixed response depending on fire burn history: it decreases over time following one fire and increases over time following two fires.

Figure 8: Average Vegetation Change (%) of Chaparral, Invasive Grass, and Coastal Sage Scrub by Time Since Second Fire

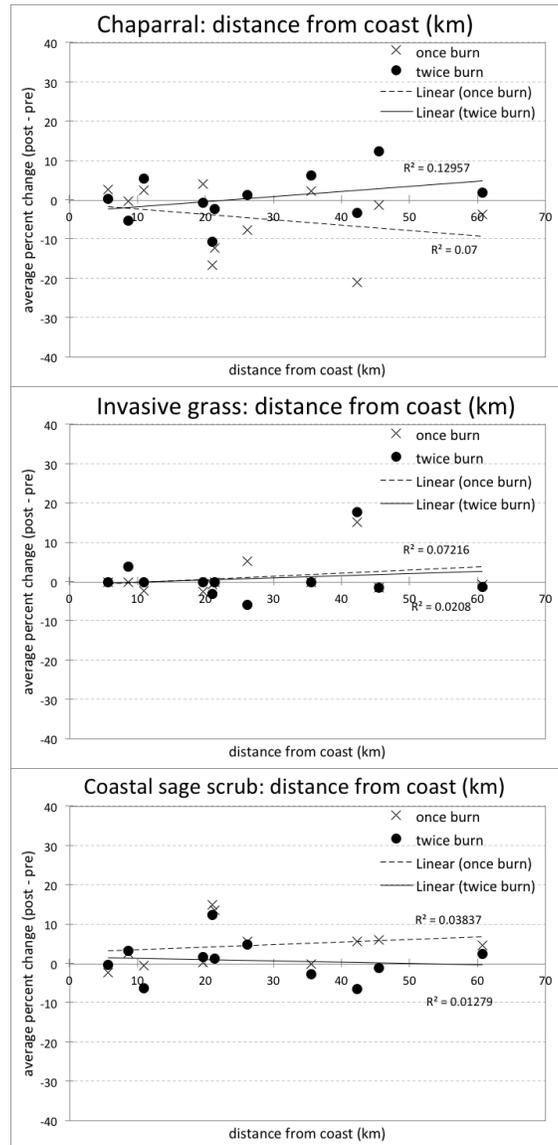


“Years since second fire” is equivalent to the regrowth years in Table 2.

To investigate if the observed percent change in vegetation cover is correlated with the “distance from coast” investigators plotted the average distance of each site to the coastline by the average percent change in vegetation cover (Figure 9). Again, “distance from coast” is not a strong explanatory variable for the observed percent vegetation change. The percent change in

invasive grass cover slightly increases with distance from the coast, the percent change in chaparral cover decreases following one fire and increases following two fires with distance from the coast, and the percent change in coastal sage scrub increases following one fire and decreases following two fires with distance from the coast.

Figure 9: Average Vegetation Change (%) of Chaparral, Invasive Grass, and Coastal Sage Scrub by Distance from Coast (km)



Average distance from coast is given in Table 2.

2.4.5 Discussion

Evidence of Southern California shrubland loss, strictly as a result of a single short interval fire, defined as two fires within five years, was not found using historical aerial photographs. No significant differences were found in the change of vegetation cover following one fire

compared to two fires in the same five year interval. Instead, vegetation change was detected when comparing north-facing and south-facing aspects. Invasive grass cover significantly increased on south-facing aspects across sites in Ventura and Los Angeles counties.

One explanation to why a large amount of variability was seen among change in vegetation cover may be due to weather and precipitation conditions the year following the second fire. Another reason may be due to the variability in species composition of each community. Chaparral as an ecoregion can have multiple compositions with varying dominant species, for example chamise dominated (*Adenostoma fasciculatum*), ceanothus dominated (*Ceanothus megacarpus*), or sumac dominated (*Malosma laurina*).

One compromise of this study was that to capture historic vegetation change (pre-1980) researchers had to use historic panchromatic aerial photographs that limited the ability to identify vegetation communities by species as is possible with many current remotely sensed data. Additionally, in order to investigate discernible large-scale patterns in vegetation change due to short interval fires, they only investigated single short interval fire events and the results may have differed if more than two consecutive fires were included in the analysis. Results also might have differed if a more complete record of a site's fire history (ex. all fires included in the FRAP dataset) were considered.

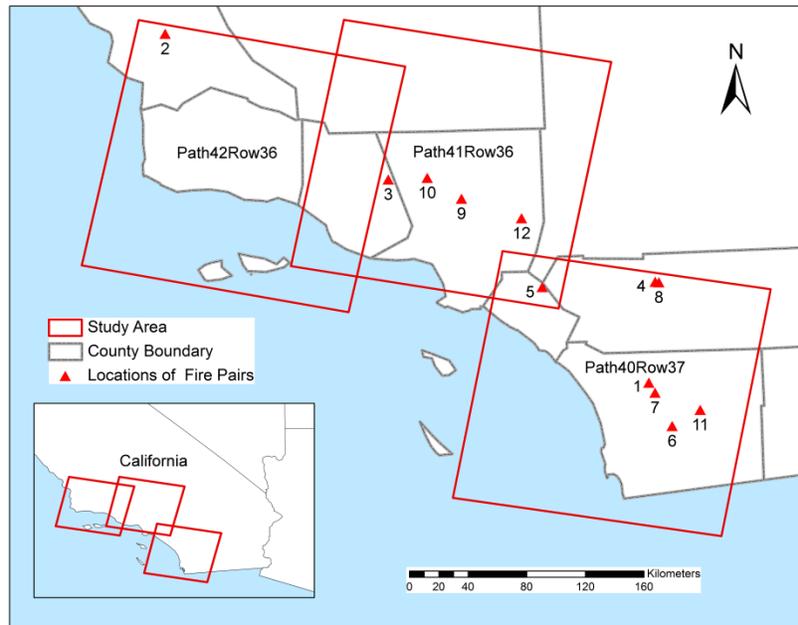
In conclusion, wide scale vegetation change was not found due to a single short interval fire event in Ventura or Los Angeles County through analyzing historic aerial photographs of paired once burn and twice burn sites from 1956 to 2003. Instead aspect was found to be a strong indicator of vegetation change, especially for invasive grasses on south-facing aspects. Thus, a single short interval fire event may not be the strongest predictor of vegetation change and, rather, other factors, such as aspect, need to be considered when predicting vegetation change in the future.

2.5 Effects of short interval fire using Landsat TM data

2.5.1 Selection of Study Region and Landsat imagery

Expanding the findings of the aerial imagery analysis to a larger geographic area required using coarser spatial resolution Landsat TM data. The study region for the landscape-scale analysis is located within the combined areas of three Landsat imagery scenes with an area of about 94,350 km² (Figure 10). According to the state and national wildfire disturbance products (e.g., Monitoring Trends in Burn Severity (MTBS), CalFire and LANDFIRE), wildfire occurred nearly 2,000 times in the study area during the 1985-2010 study period. Especially over the past several decades, this region has experienced multiple, massive fires with short interval. The combination of high fire frequency and readily available wildfire disturbance datasets make this region ideal for examining whether multiple fires in rapid succession are driving type conversion of chaparral shrublands.

Figure 10: Study Area



Source: Meng et al. 2014

The main remote sensing dataset applied to characterize effects of short interval fire on chaparral is Landsat-5 Thematic Mapper (TM) imagery covering the period from 1985 to 2010. Landsat mission imagery has over three decades record, as the world's longest continuously remote sensing data at mid-resolution, for observing land use and land change globally (Roy et al. 2014). Landsat imagery has been widely used for monitoring wildfires and studying post-fire vegetation recovery (Chu and Guo 2013; Lentile et al. 2006; White et al. 1996). Since 2010, Landsat data can be freely downloaded from USGS EROS (<http://eros.usgs.gov/>), and no restriction is applied on the usage or redistribution of Landsat data.

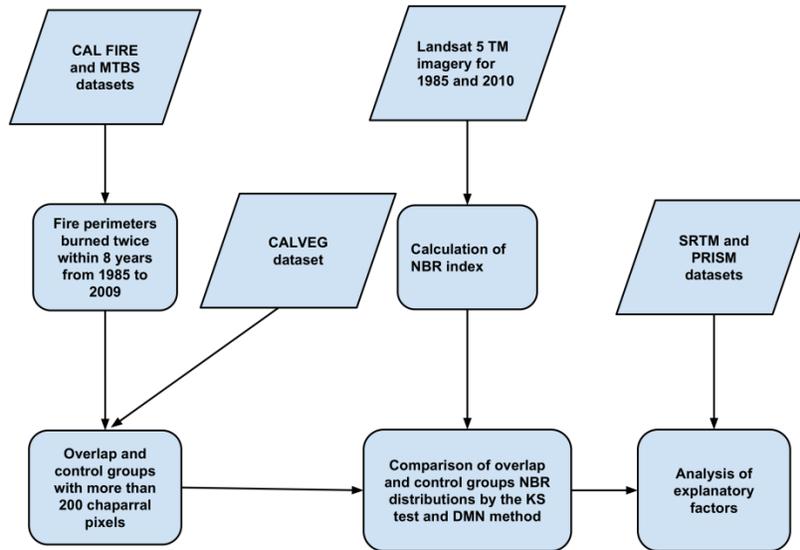
2.5.2 Change Detection Techniques

Landsat mission imagery has been long considered as suitable to detect and monitor ecological disturbances at moderate spatial and temporal scales (DeRose et al. 2011; Hurley et al. 2004; Lentile et al. 2006; White et al. 1996). Bands 3, 4, 5, and 7 from Landsat mission imagery have the largest responses to ecological disturbances, related with interactions between solar radiation and vegetation canopies (Knippling 1970; Tucker 1979; White et al. 1996). A number of vegetation indices, based on the combination of these bands, have been used successfully for monitoring and analyzing post-disturbance vegetation regrowth (Chu and Guo 2013; Gitas et al. 2012; Lentile et al. 2006). Among them, Normalized Burn Ratio (NBR), using a combination of band 4 (near infrared or NIR) and band 7 (shortwave infrared or SWIR), was not only used for identifying burned areas, but also for monitoring post-fire vegetation regrowth in a Mediterranean ecosystem (Lopez Garcia and Caselles 1991).

$$\text{NBR} = (R_{\text{NIR}} - R_{\text{SWIR}}) / (R_{\text{NIR}} + R_{\text{SWIR}}) \quad \text{Equation 1}$$

where R_{NIR} and R_{SWIR} represent surface reflectance for TM band 4 (760-900 nm) and band 7 (2,080-2,350 nm).

Figure 11: Flowchart of Methodology



MTBS is data from the Monitoring Trends in Burn Severity Project. SRTM is topographic data from the Shuttle Radar Topography Mission. PRISM is climate data from the Parameter-elevation Relationships on Independent Slopes Model.

Source: Meng et al. 2014

Based on the wildfire disturbance products and Landsat-5 TM calculated NBR, the researchers developed a methodology to detect and study the effects of short-interval fires on the chaparral within the study area (Figure 11)(Meng et al. 2014). Paired areas with two components were identified: “overlap” areas burned twice within an eight year period, and “control” areas burned just once by the second fire that burned the overlap area. Fires for both the overlap and control areas were constrained to between October 1985 and December 2009, although areas burned in the eight years prior to October 1985 were not analyzed. NBR values were calculated for both the control and overlap areas for Landsat TM images acquired pre-fire in 1985 and post-fire in 2010. Thirdly, differences in the NBR distributions between overlap and control areas were compared using two indices: D value returned by the Komolgorov-Smirnov (KS) test and NBR-based DMN (Table 3). If type conversion from chaparral to grass occurred after repeat burn, strongly positive DMN values should be expected in 2010 relative to 1985, due to reduced vegetation canopies. Variables including interval between repeat burn, elevation, post-fire temperature, and post-fire precipitation were compared to changes in NBR distributions to determine whether these variables were correlated with vegetation recovery and potential type conversion.

Table 3: Descriptions of Indices Used in This Study

Indices	Calculations	Indications
D_{1985}	The maximum difference between overlap and control groups in NBR cumulative distributions from the 1985 image	Difference in vegetation cover between a control area and a corresponding overlap area in 1985
DMN_{year}	Median (Control(NBR_{year})) – Median (Overlap(NBR_{year}))	Higher (positive)/lower (negative) vegetation cover within a control area relative to a corresponding overlap area in a specific year
Change in DMN	$DMN_{2010} - DMN_{1985}$	Increase (positive)/decrease (negative) in vegetation cover within a control area relative to a corresponding overlap area after repeat burn

Source: Meng et al. 2014

Vegetation cover in overlap and control areas was validated using high resolution imagery from 2009-2013 in Google Earth. One hundred points were randomly generated within the overlap and control areas of each targeted fire pair. Image interpretation was used to assign each randomly generated point as having evergreen shrub canopy or non-evergreen shrub canopy (e.g., grass, drought-deciduous shrub, soil). Lastly, differences in percentages of evergreen shrub canopy pixels between the control and overlap groups were compared to DMN_{2010} .

2.5.3 Results

A total of 39 potential overlap and control pairs were identified within the study area during the 1985-2009 period. Since this study focused on the potential effects of short-interval repeat burn on the chaparral, only the 12 fire pairs with similar 1985 pre-fire NBR distributions were further compared (Table 4). A threshold D_{1985} value of 0.16 was used for selecting overlap and control pairs with similar 1985 pre-fire NBR distributions.

Table 4: Comparison of Results for Fire Pairs with Similar Cumulative Distributions of 1985 NBR Values^a

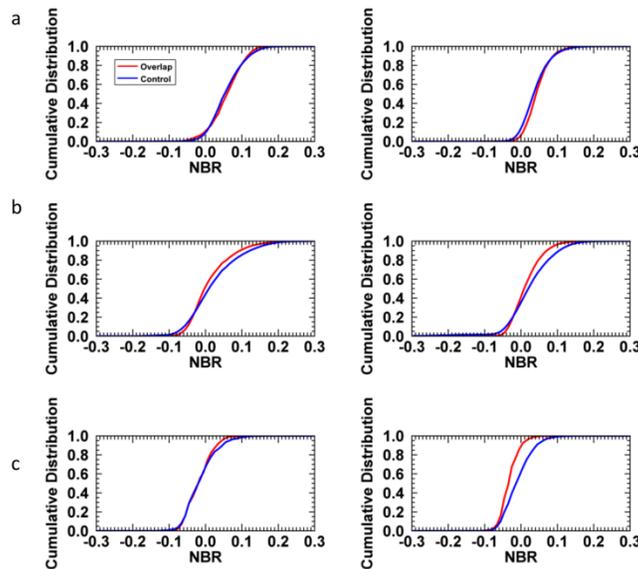
ID Number	Path Row	Year1 Fire1	Year2 Fire2	D ₁₉₈₅	Change in DMN (2010-1985)	Overlap Area (ha)
1	p40r37	2003 Paradise	2007 Poomacha	0.135	0.045	1031
2	p42r36	1996 Highway58	2003 Parkhill	0.078	0.037	40
3	p40r37	2003 Piru	2007 Ranch	0.065	0.025	79
4	p40r37	2001 Silent	2006 Esperanza	0.140	0.017	22
5	p40r37	2002 Green	2006 Sierra	0.128	0.016	214
6	p40r37	2003 Cedar	2007 Witch	0.056	0.009	2130
7	p40r37	2003 Paradise	2007 Witch	0.057	-0.001	1518
8	p40r37	1999 Pine	2006 Esperanza	0.066	-0.005	210
9	p41r36	2007 North	2009 Station	0.086	-0.018	60
10	p41r36	2002 Copper	2007 Buckweed	0.060	-0.019	1368
11	p40r37	1999 Banner	2002 Pines	0.110	-0.051	23
12	p41r36	1996 Bichota	2002 Curve	0.094	-0.095	25

^a Sorted by decreasing change in DMN. ID number refers to fire pairs shown in Figure 10. A more positive change in DMN would be consistent with type conversion, however, change in DMN was neither strongly positive or negative for the NBR distributions examined.

Source: Meng et al. (2014)

These fire pairs demonstrated different responses to short-interval repeat burn (Figure 12). For example, after repeat burns, some fire pairs' NBR distributions for the overlap area shifted to higher median NBR than the control area (Figure 12a); some fire pairs' differences between control and overlap NBR distributions became smaller (Figure 12b); and a few fire pairs showed that the overlap area shifted to lower median NBR relative to the control area (Figure 12c), as would be expected for type conversion.

Figure 12: Pre-fire (1985, left column) and Post-fire (2010, right column) Cumulative Distributions of NBR Values from Different Fire Pairs



- a. 1999 Pine fire and 2006 Esperanza fire b. 2003 Cedar fire and 2007 Witch fire.
c. 1996 Highway 58 fire and 2003 Parkhill fire.

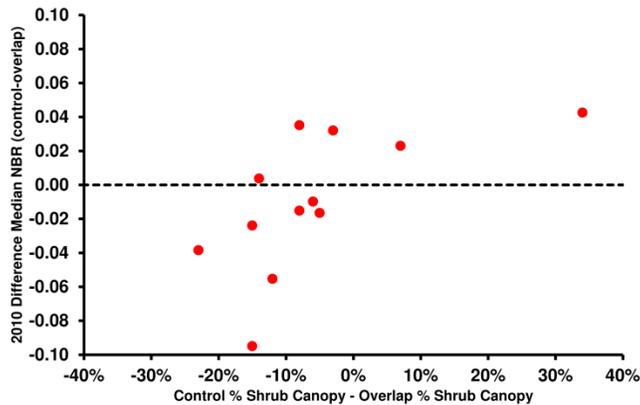
Source: Meng et al. (2014)

Contrary to expectation, substantial changes in DMN did not exist in most of 12 fire pairs. Six of twelve cases with a total area of 3515 ha (52% of the burned area evaluated) have a positive change in DMN (Table 4). The other cases have a negative change in DMN. However, most of the cases' change values were relatively small, signifying relatively small changes in chaparral cover after the repeat burn.

A significant, positive correlation (Pearson's correlation: 0.63, p-value: 0.03) existed between DMN_{2010} and difference in percent evergreen shrub canopy cover calculated from Google Earth (Figure 13). It indicated that differences in DMN, calculated from 30 m Landsat-5 TM data, were correlated with differences in evergreen shrub cover between control and overlap areas on the ground.

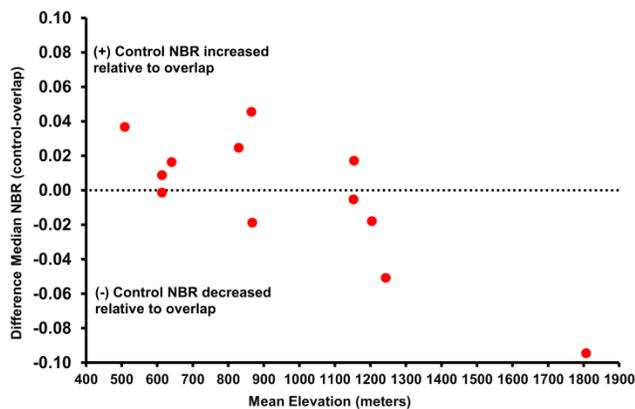
No apparent trend between change in DMN and variables such as time interval between fires, recovery time, mean post-fire precipitation in wet season or mean minimum temperature in January was detected. By contrast, elevation was the only strong factor (Pearson's correlation: -0.80, p-value: 0.0019) for explaining the variation in DMN following short-interval fires within the study area in the 1985-2009 period (Figure 14). This trend indicates that reduced vegetation cover following short-interval fires is more likely to occur at lower elevation than at higher elevation.

Figure 13: Post-fire Difference in Percent Evergreen Shrub Canopy, Derived from Google Earth Imagery for Control and Overlap Areas, versus DMN2010



Source: Meng et al. (2014)

Figure 14: Mean Elevation versus Change in DMN.



Source: Meng et al. (2014)

2.5.4 Discussion

Extensive, landscape-scale type conversion following recent short-interval fires was not detected by the remote sensing-based approach. Positive changes in DMN, as a proxy for reduction in vegetation cover at the repeat burned areas, were found for 6 out of 12 sites examined. However, only two of these six overlap/control areas, accounting for 16% of the burned area evaluated, showed large changes in DMN, consistent with type conversion. These two fires were both at lower elevation, supporting a trend that implies that vegetation cover may be more impacted by short-interval fires at lower elevations. Our results support the findings from previous studies on rangeland improvements, that it is difficult to convert existing chaparral to grassland (Bentley 1967; Fuhrmann and Crews 2001). Type conversion of shrubland to grassland may be a spatially limited and regionally rare phenomenon currently (Meng et al. 2014).

Among exploratory variables, elevation was the only strong predictor for explaining variations in shrubland recovery following short-interval fires. This finding can be explained by the fact that non-sprouting species decrease with elevation (Hanes 1971) in southern California chaparral shrublands, and multiple burns usually affect non-sprouting species more significantly related with its physiological recovery mechanism, compared to sprouting species (Keeley and Brennan 2012; Zedler et al. 1983). Another possibility is that the vulnerability of low elevation sites was higher compared to high elevation sites, due to human activities, such as urbanization and increased ignition frequency.

However, we recognize some limitations existing in the current study. Due to the relatively coarse spatial and spectral resolution of Landsat TM imagery, this study cannot rule out changes in chaparral species composition following short-interval fires. In addition, cumulative, small-scale type conversions are beyond the capacity of Landsat-based analysis, and this analysis did not include areas burned more than twice. The study also could not detect locations sensitive to long-term type conversion and already converted many decades ago. However, any systematic change was not detected in the qualitative shape of NBR distributions (Figure 12).

In conclusion, based on historical wildfire datasets and remote sensing imagery, differences in vegetation cover following short-interval fires were analyzed in southern California chaparral shrublands during the 1985-2009 period. Changes in the distributions of NBR values from paired overlap/control fire pairs did not find evidence of extensive type conversion of shrubland to grassland, suggesting that type conversion of shrubland to grassland was likely to be a spatially limited and regionally rare phenomenon in the study period.

2.6 Ecophysiological responses of three dominant chaparral species to rainfall and extreme drought in Southern California

2.6.1 Introduction & rationale

A diversity of chaparral shrubs has developed physiological adaptations to tolerate both seasonal and occasionally severe drought. Yet if drought is severe enough, some species may die and chaparral composition may shift to favor those species with adaptations to continue to photosynthesize under extreme conditions. Indeed, several studies have demonstrated differential tolerance of chaparral species to drought (e.g., Kolb and Davis 1994, Jacobsen et al. 2007a,b, Pratt et al., 2007, 2014).

To understand the relative vulnerability of dominant chaparral species to the effects of seasonal (and extreme) drought, this study evaluated variation in photosynthesis and water use among individuals of three dominant species in a heterogeneous stand of mesic mixed chaparral over two summer drought periods. These happened to be years of extreme drought (NOAA, www.droughtmonitor.unl.edu). The species studied represented two species that resprout from basal lignotubers after fire and one species that is an obligate seeder. Several studies (Jacobsen et al. 2007a, Pratt et al. 2007) suggest that seeding species should be more tolerant of drought than sprouting species because their survival depends on aboveground survival. To prevent the die-off of aboveground reproductive branches, seeding species would need to be more drought tolerant. Hence this study evaluated how representatives of these three important chaparral

genera including a seeder and two sprouter species, tolerated extreme drought conditions and the extent to which water use by chaparral plants was curtailed by drought. The physiological values obtained will then be used to evaluate landscape scale water use as described below.

The study addressed the following specific questions and hypotheses:

1. How do carbon gain (photosynthesis) and water loss (transpiration), as controlled by stomatal conductance, differ between three dominant chaparral shrub species as plant-available water changes seasonally?

The hypothesis here was that the obligate seeding species would maintain higher photosynthesis as the summer drought ensued because of its evolutionary history and reliance on survival between fire events. Alternatively, because sprouter species have reserves in basal burls, they may show greater ability to photosynthesize as the summer drought ensued.

2. How do chaparral shrub species differ in their capacity to photosynthesize during seasonal dry periods as well as during exceptional drought?

Here species responses to initial drought were compared in the first year versus final responses at the end of the second year (exceptional drought). The researchers predicted that responses to the initial summer drought would differ among species, but by the end of the field season in 2014, all species would have completely shut down and converged in leaf water potential and gas exchange due to the extreme conditions.

The ecophysiological relationships that were derived from these studies were integrated into an ecohydrologic model, RHESSys, to improve estimates of streamflow in an already drought-prone region of southern California. See section 4.4 for related ecohydrologic modeling results.

2.6.2 Southern California Ecophysiological Study Site and Species

Most chaparral plant communities in California occupy habitats that are highly heterogeneous in topographic relief and soil characteristics – two important environmental factors that control water availability to plants. The Santa Barbara region, in particular, supports a high diversity of chaparral species relative to other regions in California due to the variety of microhabitat and climate conditions driven by complex interactions between geologic features and high intra- and inter-annual variability in rainfall.

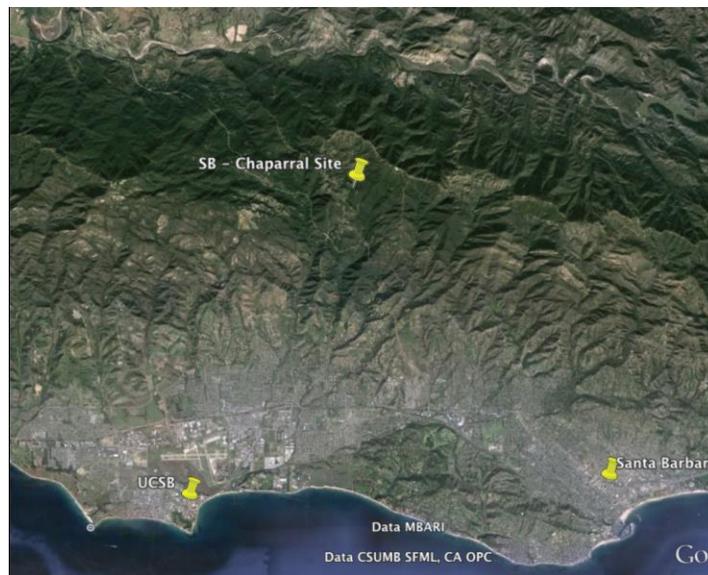
The study site was a stand of mixed chaparral at 900 m in elevation on a generally south to southwest facing slope in the Santa Ynez Mountains, near Santa Barbara, California (Figure 15). The substrate consists of uplifted sandstone of the Coldwater Formation from the Eocene Age. Soils are sandstone derived and poorly developed due to the rockiness of the terrain. The site was last burned in 1965, and is dominated by mature chaparral species, particularly species in the genera *Ceanothus*, *Arctostaphylos*, and *Adenostoma*. The long-term average rainfall at this study site is 877 mm per year (www.countyinfosb.org/pwd).

The study was initiated during the water year (WY) 2013 and completed in the WY 2015. California water years are measured from October 1 – September 30 of the following year to account for the fact that most rainfall occurs in the winter. Initial field measurements were

made in May 2013, and the final field measurements were conducted in December 2014. Sampling was repeated roughly every three weeks as weather allowed. Precipitation during the 2012-2013 calendar years was 52% (456 mm) of the 50-year average, while during the 2013-2014 year it was 35% (306 mm) of the average. These two drought years were preceded by another year of lower than usual precipitation, in which precipitation was 54% (474 mm) of normal, making this a dramatic three-year drought.

The three dominant chaparral species were selected for study: Greenbark ceanothus (*Ceanothus spinosus*), Chamise (*Adenostoma fasciculatum*), and Big berry manzanita (*Arctostaphylos glauca*) (n=7 per species). The former two are post-fire re-sprouters, while the latter is an obligate seeding species.

Figure 15: Field Site in Santa Barbara Foothills



2.6.3 Methods

The study compared leaf-level ecophysiological responses of these three dominant chaparral shrub species to changes in seasonal water availability between May 2013 and December 2014. The sampling period spanned two summer dry periods and one wet winter. The entire twenty-month monitoring period between May 2013 and December 2014 fell within the definition of a 'severe' to 'exceptional' drought according to the United States Drought Monitor (Drought Monitor, <http://droughtmonitor.unl.edu/>; NOAA).

2.6.3.1 Quantifying seasonal variation in plant water availability and use

The researchers assessed changes in available water to plants at their rooting zone by measuring predawn (0200-0400 hr) leaf water potential (Ψ_{pd} , measured in megapascals, MPa) using a Scholander pressure chamber (Model 1000, PMS Instruments Inc., Albany, OR). When Ψ_{pd} values are close to zero, plants are more hydrated than if values are larger and more negative. At roughly three-week intervals for the duration of the study, Ψ_{pd} values were measured from

one to two twigs collected from each individual plant. One caveat is that in 2013, the pressure chamber could only record Ψ_{pd} values up to -6.5 MPa; however, several shrubs reached Ψ_{pd} values more negative than this limit by August 2013. Thus, the minimum (most negative) Ψ_{pd} value for species during summer 2013 is not known. Beginning in May 2014, they obtained a pressure chamber from which they could sample minimum Ψ_{pd} value over the year.

2.6.3.2 Shrub Characteristics

In the winter (February 14th) and summer (June 17th) of 2014, the leaf area index (LAI) of each shrub was measured using a Licor LI-2000. LAI measurements were collected just after dawn for each individual to characterize any changes in canopy leaf area between the wet and dry season.

2.6.3.3 Photosynthesis & Stomatal Conductance

Investigators coupled measurements of Ψ_{pd} with leaf gas-exchange rates using a Licor 6400 Portable Photosynthesis system. They collected measurements from one leaf per individual per species at four different time blocks through the day (0800, 1100, 1400, and 1600 hr) to track how photosynthetic rates would change over the course of a day as well as a season. They report the leaf gas-exchange rates observed during late morning (between 1030-1200 hr) because this is when maximum rates were typically observed. They repeated LiCOR measurements on the same leaf for each measurement, if the leaf was in good condition. Measurement time and environmental conditions around the leaf were controlled to match the average local conditions on each sampling date. For example, during winter, leaf temperature was 14°C and photosynthetically active radiation (PAR) was set to 600 nm. In the summer, leaf temperature was 29°C and PAR was 1800 nm.

2.6.3.4 Data Analysis

Researchers calculated the average Ψ_{pd} and stomatal conductance (g_s) per individual from each sampling date. A repeated measures ANOVA was performed to test for statistical differences in ecophysiological responses between chaparral shrub species over time. If significant ($P < 0.05$), they performed a pairwise comparisons to identify points in time when these functional groups differed significantly from one another using JMP version 10.0.0 software (SAS Institute, Cary, SC). To assess the strength of the relationship between changes in plant water availability (Ψ_{pd}) and use (g_s) for the different species, they first normalized average g_s for each sampling period by the seasonal maximum and plotted these values against average predawn leaf water potential. The best-fit line to these data was generated in Excel.

2.6.4 Results

Total rainfall during WY 2013 and WY 2014 was below average for the study location (Figure 16). Cumulative rainfall in WY 2013 totaled 382 mm, and arrived in several small pulses in the winter and early spring. WY 2014 received a total of 259 mm of rainfall, with two main pulses of water coming in November and late February.

The LAI of all species remained constant from winter to mid-summer (Table 5). *A. fasciculatum* and *A. glauca* had higher average LAI values than *C. spinosus* throughout the experiment.

Table 5: Average LAI Values for Shrub Species in Winter and Summer

Species	Average LAI ± Standard Error	
	February 14 th , 2014	June 17 th , 2014
<i>Adenostoma fasciculatum</i>	1.49 ± 0.11	1.58 ± 0.27
<i>Arctostaphylos glauca</i>	1.58 ± 0.14	1.60 ± 0.15
<i>Ceanothus spinosus</i>	1.09 ± 0.16	1.07 ± 0.17

Ψ_{pd} decreased for all species throughout the summer drought, and increased with winter rainfall events in both years (Figure 17). There was a significant difference between species ($p < 0.0001$) over time.

Figure 16: Total Monthly Rainfall in Santa Barbara Foothills through Duration of this Study

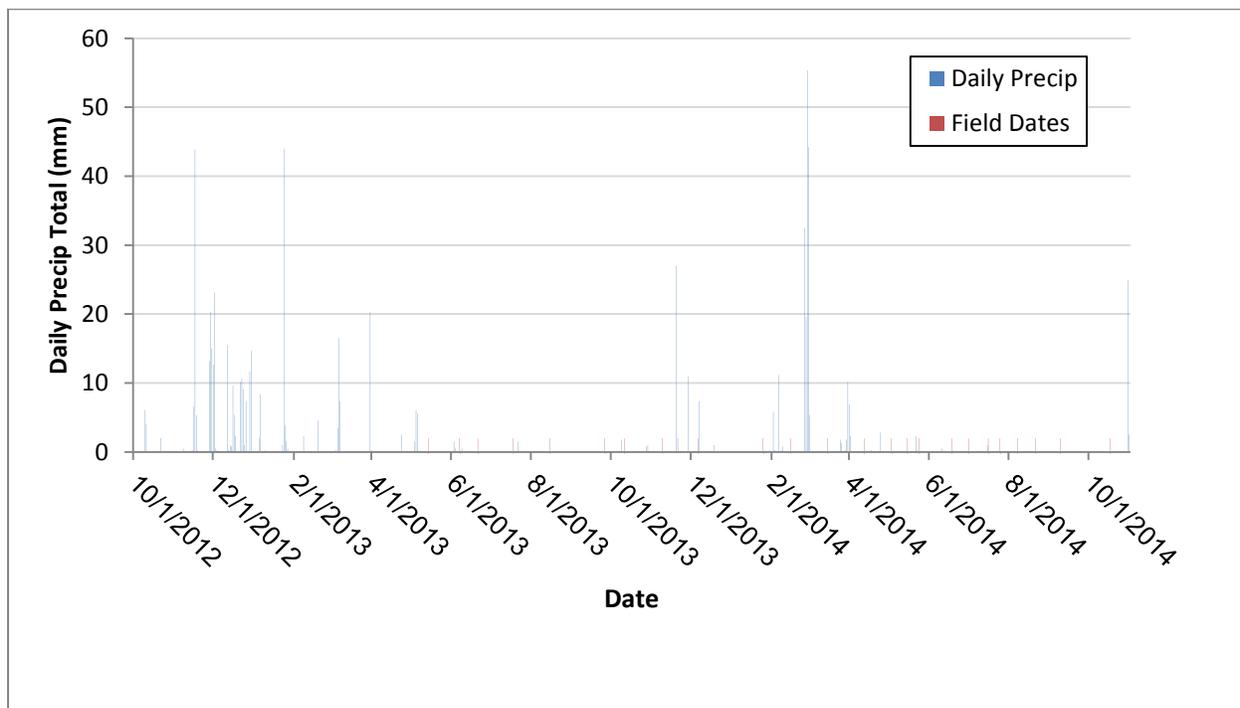
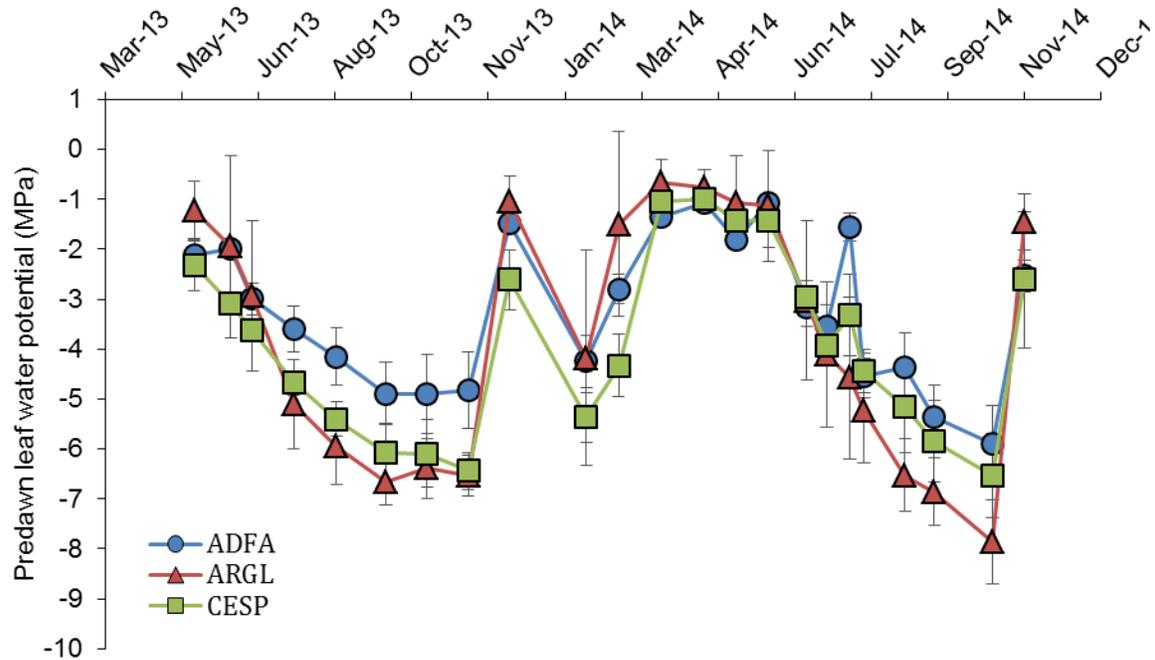


Figure 17: Average Ψ_{pd} at Each Sampling Data for Three Dominant Chaparral Species (n=7 per species) Through Time. Error Bars Indicate \pm SE

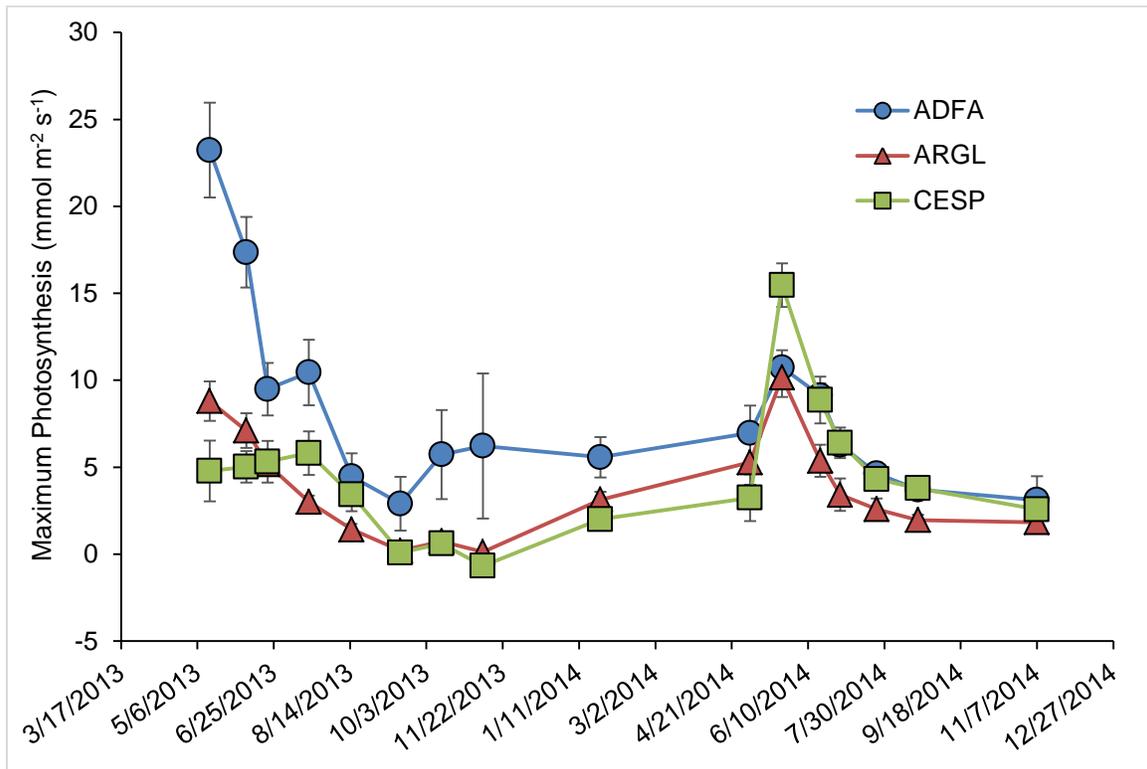


ADFA = *Adenostoma fasciculatum*; ARGL = *Arctostaphylos glauca*; CESP = *Ceanothus spinosus*

In 2013, there was a significant difference between the Ψ_{pd} of species over time, with *A. fasciculatum* having a significantly higher (less negative) Ψ_{pd} throughout most of the summer and fall (Figure 17, $p=0.0002$). The minimum (most negative) Ψ_{pd} ever reached by *A. fasciculatum* was approximately -5 MPa, whereas both *A. glauca* and *C. spinosus* reached average Ψ_{pd} values as negative as -6.5 MPa. In 2014, the differences between the Ψ_{pd} values of the three species was only marginally significant (Figure 17, $p=0.606$).

Photosynthetic rates did not increase as rapidly or dramatically as Ψ_{pd} values did following winter rains in either 2013 or 2014 (Figure 18). Species differed in their photosynthetic rates in 2013, with *A. fasciculatum* often having slightly higher photosynthetic rates than either other species (Figure 18). Photosynthetic rates were more similar between species in 2014 (Figure 18).

Figure 18: Average Photosynthetic Rates of Three Chaparral Shrub Species \pm SE



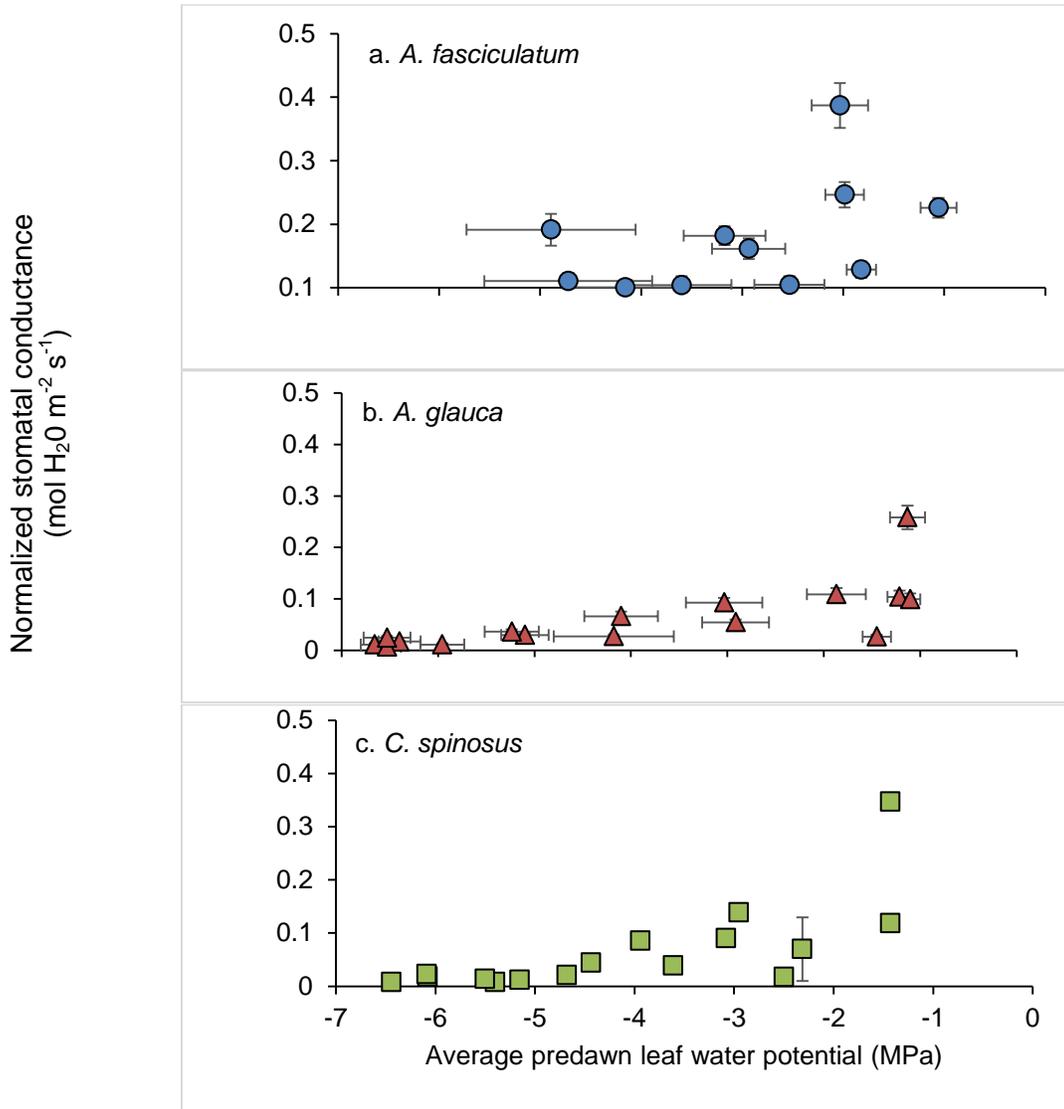
ADFA = *Adenostoma fasciculatum*; ARGL = *Arctostaphylos glauca*; CESP = *Ceanothus spinosus*

There was a significant difference between species in 2013 ($p=0.0002$), and marginal significance between species in 2014 ($p=0.0606$).

Average g_s was positively correlated with average photosynthesis for all three species (*A. fasciculatum* $R^2=0.86$; *A. glauca* $R^2=0.93$; *C. spinosus* $R^2=0.26$); therefore, patterns of photosynthesis can be used as a proxy for stomatal conductance.

For all species, the ratio of average stomatal conductance normalized by maximum rates increased as xylem pressure potential increased following a power law relationship (Figure 19).

Figure 19: Relationship Between Average Normalized g_s and Ψ_{pd} for Each of the Chaparral Species. Error Bars Represent \pm SE



2.6.5 Discussion

Although all three chaparral species examined in this study are believed to be adapted to typical summer drought, their ecophysiological responses varied during the exceptional drought that occurred during the study. Very little rain fell over two consecutive winters, which led to low water availability for all species during the summers (Figure 16, Figure 17). *A. glauca* and *C. spinosus* both reached Ψ_{pd} values below their published Ψ_{P50} values (-4.67 MPa and -4.68 MPa, respectively), indicating that the extended drought period may have caused some level of xylem embolism in these species (Choat 2012). Conversely, *A. fasciculatum* did not reach a Ψ_{pd} below its Ψ_{P50} of -7.98 MPa, suggesting that its xylem integrity may have been less affected by the drought conditions during the study period than that of the other shrub species (Choat 2012).

All three species had lower water availability in the late summer and fall than in the spring, but *A. fasciculatum* consistently had a less negative Ψ_{pd} values in 2013, indicating higher water availability throughout the dry season. *A. fasciculatum* is believed to have well-developed, deeply penetrating rooting systems, whereas both *A. glauca* and *C. spinosus* are thought to have spreading, shallower roots (Hellmers 1955, Thomas and Davis 1989). The greater rooting depth of *A. fasciculatum* may allow it to access deeper, alternate water sources that cannot be reached by the other two species.

The lack of change in LAI between the winter and summer suggests that dropping leaves was not a mechanism used to decrease transpirational water stress by any of the species (Table 5). It is also possible that leaf development was reduced due to the dry conditions, or that the timing of the measurements missed the period of highest LAI.

Rainfall events appear to have important effects on the water availability of all three chaparral shrub species (Figure 17). The rapid increase in Ψ_{pd} values (from more to less negative values) after the December 6th, 2013 rainfall event (7.4 mm) indicates that all three species were able to access the rain immediately after the event (Figure 16, Figure 17). However, all three species subsequently also decreased in their Ψ_{pd} values again by January, indicating that they are able to take advantage of smaller rainfall events, such as those that occurred between January 29th, 2014 and February 11th, 2014 (18.5 mm total). *A. glauca* responded particularly strongly to these small rain events, as indicated by its more dramatic relative rise in Ψ_{pd} . The increase in Ψ_{pd} values in all species lasted several months into the spring, indicating that even small, pulsed rain events during and after drought periods can have lasting effects for these species.

The trend seen in Ψ_{pd} values over time is mirrored in the maximum photosynthesis rates of all three species (Figure 18). Maximum photosynthesis rates increased after rainfall events, though the magnitude of the increase was not always the same for each species. There is a clear and strong relationship between Ψ_{pd} values and normalized stomatal conductance (Figure 19). Not unexpectedly, stomatal conductance, which correlates well with photosynthetic rates, decrease as water availability decreases. Furthermore, both *A. glauca* and *C. spinosus* appear to have closed their stomata when their Ψ_{pd} values reached -6.0 MPa. On the other hand, *A. fasciculatum* appears to have never reached a low enough Ψ_{pd} values to induce stomatal closure, suggesting that this species could continue photosynthesizing even when water stress was too great for the other species. As described earlier, *A. fasciculatum* may be able to access a water resource not available to the other species. Additionally, *A. fasciculatum* may be more drought tolerant overall than the other two species in the study, as it appears to have higher normative photosynthesis rates at lower Ψ_{pd} values than either *A. glauca* or *C. spinosus*.

The results of the monitoring suggest that these shrubs will slow or stop photosynthesizing when their access to water is limited for a long period of time, as shown by the minimal levels of photosynthesis in 2013 and 2014. To reduce the likelihood of developing catastrophic xylem embolisms, these shrubs may close their stomata and remain semi-dormant until a future pulse of rain. Continued persistent drought could lead to carbon starvation if plants are unable to open their stomata without risking hydraulic failure. However, sporadic and small rain events may sustain these shrub species through even extended, exceptional drought conditions,

suggesting that many plants will be able to recover normal hydraulic function when drought conditions end.

CHAPTER 3:

Sierra Nevada Ecosystem Response to Fire and Drought

3.1 Rationale & Changing Fire Regimes in Conifer Forests

There is widespread concern that both climate change and altered fire regimes will lead to vegetation change across California, which in turn will influence runoff amounts, stream flows and hydropower generation. Depending on the location, an increase in ambient temperatures, wildfire occurrence and disease incidence plus a reduction in precipitation will likely lead to altered native vegetation distribution also affecting the amount of woody biomass available for bioenergy production. In the Sierra Nevada Mountains of California, this vegetation type conversion is likely to be from forest to shrubland. Although plant cover is known to affect basin hydrology and watershed nutrient cycling, sensitivities to vegetation type (forest versus shrub versus grass) and spatial arrangement of species within watersheds are not well understood. A key objective of this project was to evaluate the degree to which vegetation composition, and type conversion from current distributions, may influence stream flow, hydropower generation from the Sierra Nevada Mountains. Specifically, parameterization of an eco-hydrological model using field-based plant ecophysiology measurements (plant water status and leaf gas exchange measurements) in representative watersheds of these regions should result in a more complete understanding of how vegetation treatment (or restoration) efforts can meet watershed management goals, including energy production.

In addition, forest ecosystems in the Sierra Nevada Mountains are greatly influenced by wildfire as a natural disturbance, yet increasing occurrence of fire and drought may alter the course of post-fire recovery in these systems. Analysis of the controls over vegetation response following fire is essential for sustainable management of the multiple resources valued in these vast ecosystems, particularly given climate trends projected for the Sierra Nevada Mountains. Due to anticipated impacts of climate change on western US ecosystems, there is a considerable body of research concerning how climate change has and will impact forest structure, disturbance regimes, and carbon storage and dynamics (Allen et al. 2010; Bond-Lamberty et al. 2014; Dale et al. 2001; Dennison et al. 2014; Rogers et al. 2011). Wildfire has a key role in shaping patterns and processes in terrestrial ecosystems of the western United States. Fire-prone environments have promoted the adaptation of vegetation species in many western United States forests, enabling these fire-adapted species to recover following fire (Sugihara et al. 2006). Nevertheless, post-fire forests regeneration dynamics are still determined by many onsite factors, such as fire severity, plant regeneration strategies, topography and local climate.

These studies have made substantial contributions to our understanding of vegetation dynamics following fire in the Sierra Nevada Mountains, but most of them are ground-based and face logistical limitations in the extent of the areas that could be sampled. Therefore, variations within and among studies are likely to be common and large sample sizes are needed to develop predictive relationships that will apply over larger scales. By contrast, remote

sensing based post-fire studies can provide insights into landscape-scale controls over vegetation responses to wildfire beyond what can be measured in spatially limited ground surveys.

Remote sensing has been widely considered to be suitable for monitoring forest fires, as well as for post-fire vegetation recovery (Chen et al. 2011; Chu and Guo 2013; Díaz-Delgado and Pons 2001; Gitas et al. 2012; Lentile et al. 2006; Mitri and Gitas 2013; White et al. 1996). Changes in spectral and spatial properties of forests caused by wildfires enable use of remote sensing for such purposes (Vicente-Serrano et al. 2011; White et al. 1996). Spectral vegetation indices, based on a ratio of reflectance in visible, near infrared, and shortwave infrared bands, have been most extensively examined to monitor and quantify post-fire vegetation dynamics (Ireland and Petropoulos 2015; Lentile et al. 2006). Imaging spectroscopy and LiDAR have more recently been incorporated into investigations of post-fire vegetation recovery, in terms of species composition or vertical structure (Huesca et al. 2013; Kane et al. 2015; Kane et al. 2013a; Kane et al. 2013b). However, spatial and temporal coverage of the new sensors limited the areas or time period they examined.

Land managers endeavor to identify vulnerable areas with poor post-fire recovery potentials, and then restore natural forest conditions with priority intervention (Collins et al. 2007; Collins et al. 2009; Huesca et al. 2013). Thorough and accurate monitoring, evaluation and understanding of post-fire forest regeneration are essential for assessing effects of disturbances on ecological processes, modeling vulnerability of forest ecosystems, and studying climate-fire regime interactions (Mitri and Gitas 2013; Solans Vila and Barbosa 2010). Using forest gap and climate models, optical remote sensing, and LiDAR data, a number of studies have recently explored the effects of local climate patterns and topography on fire regime and forest structure in the Sierra Nevada Mountains in depth (Kane et al. 2015; Kane et al. 2013a; Kane et al. 2013b; Miller and Urban 1999a, b). However, to our knowledge, few studies have investigated the effects of fire severity and post-fire climate with topographic factors on vegetation dynamics across a large spatial-temporal scale.

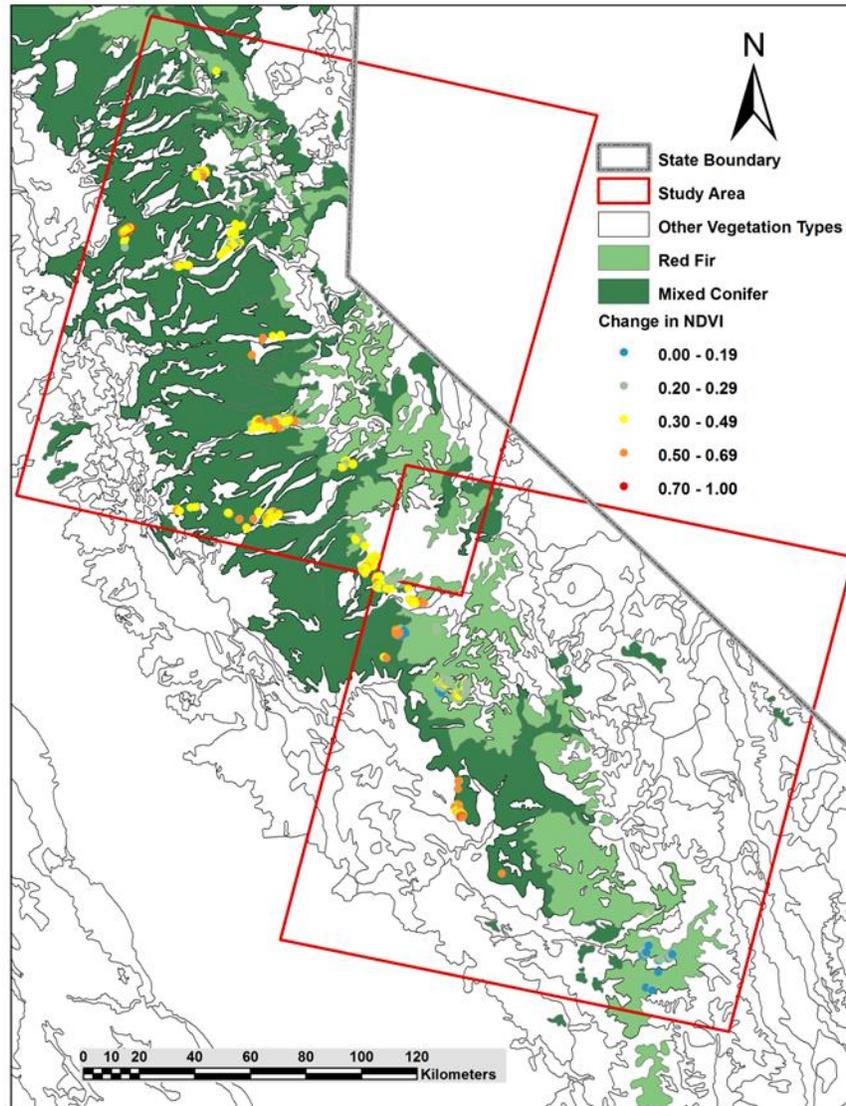
3.2 Effects of Fire Severity and Post Fire Drought on Vegetation Recovery

3.2.1 Selection of Study Region and Landsat Imagery

Using times series Landsat 5 TM data (1994-2011), the study compared landscape-scale effects of fire severity and post-fire climate with topographic factors (e.g., elevation, slope, aspect) on short-term vegetation regrowth of mixed-conifer and red fir forests in the Sierra Nevada. The study area encompassed the central portion of the Sierra Nevada mountain range, with two Landsat tiles containing an area of about 46,500 km² (Figure 20). Within the study area, vegetation varies with elevation from chaparral shrubland communities at lower elevation (~380 to 1,500 m), mixed conifer forests at mid-elevations (~1,100 to 1,900 m), lodgepole pine and red fir forests (~2,400 to 3000 m) to subalpine forests and alpine meadows at the higher elevation (above around 2,650 m) (Storer and Usinger 1963). We focused on mixed-conifer and red fir forests as mapped by the 1977 Classification and Assessment with Landsat of Visible Ecological Groupings project (1977 CALVEG), which provides a pre-fire snapshot of vegetation

type. Mixed-conifer forests primarily consist of white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), sugar pine (*Pinus lambertiana*), ponderosa pine (*P. ponderosa*), Jeffrey pine (*P. jeffreyi*), incense-cedar (*Calocedrus decurrens*), California black oak (*Quercus kelloggii*), and other hardwood species (Collins and Roller 2013). Mean fire return interval generally increases with elevation, which is a characteristic of Sierra Nevada ecosystems (Swetnam et al. 1998).

Figure 20: The Sierra Nevada Remote Sensing Study Area



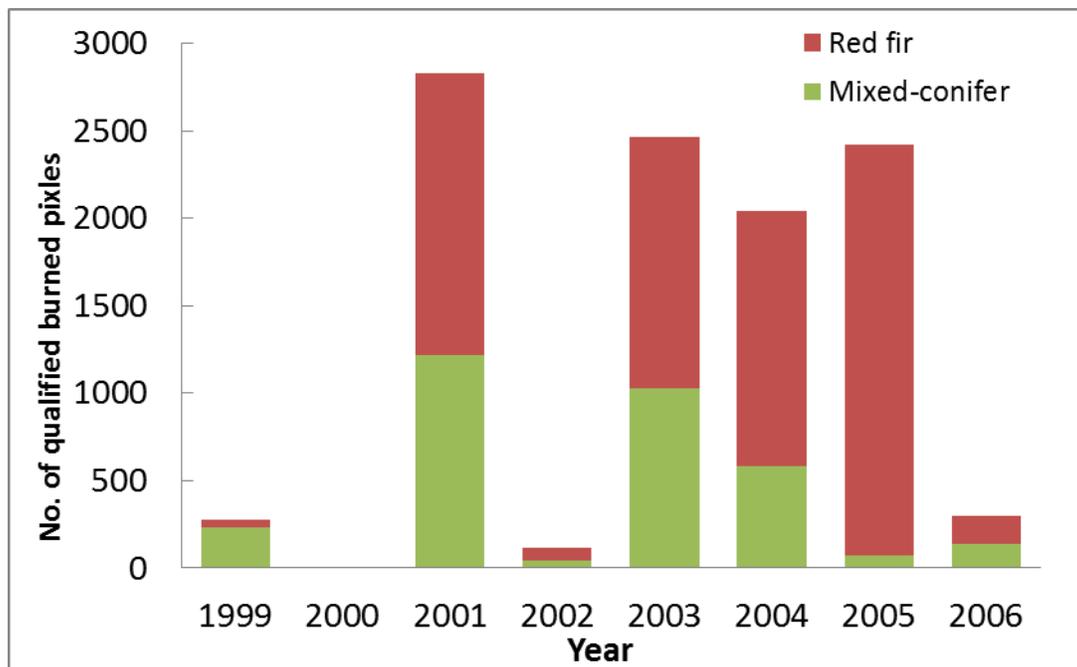
Map includes areas classified as mixed conifer and red fir in the 1977 CALVEG dataset and the extent of two Landsat scenes used in the study. Points show change in Normalized Difference Vegetation Index (NDVI) between one year following fire and five years following fire, indicating vegetation recovery.

3.2.2 Analysis Techniques

The main dataset applied to characterize vegetation recovery following fire was based on Landsat Time Series Stacks (LTSS), covering the period from 1994 to 2011. LTSS have been preprocessed for radiometric normalization and masking of cloud and cloud shadow (Huang et al. 2010a; Huang et al. 2010b). Corresponding maps of forest disturbance history (1984 to current) were generated by Vegetation Change Tracker (VCT) in 30 m spatial resolution consistent with LTSS. VCT is a remote sensing automated algorithm for reconstructing forest disturbance history (Huang et al. 2010a). Based on the spectral-temporal properties of land cover and forest change processes, VCT disturbance maps record the changes in forests occurring over a particular year, while not providing detailed information on the disturbance type (Huang et al. 2009; Huang et al. 2010a). MTBS (Monitoring Trends in Burn Severity; <http://www.mtbs.gov/>) data were used to supplement the VCT disturbance maps. Other data used for analysis included 30 m elevation, slope, and aspect derived from a DEM, and 4 km gridded monthly mean precipitation, January minimum temperature, and July maximum temperature from PRISM (Daly et al. 1994, <http://www.prism.oregonstate.edu>).

We examined large fires (> 405 ha) occurring between 1999 to 2006 within the study area, and analyzed the vegetation cover 5 years before and up to 6 years after the fires (1994-2011). Pixels burned once during the 1999-2006 period according to VCT disturbance and the MTBS dataset, and that belonged to the mixed conifer or red fir classes in the 1977 CALVEG dataset were selected. Pixels detected as disturbed prior to 1994 or having been disturbed more than once by VCT were excluded from the analysis. The number of qualified burned pixels after masking procedures varied from year to year: no qualified pixels were available in 2000, whereas in 2001, 2003, 2004 and 2005 there was a large number of qualified pixels (Figure 21).

Figure 21: Number of Qualified Burned Pixels in Each Year of the Landsat TM Time Series



For each year in the time series, Normalized Difference Vegetation Index (NDVI) was calculated using Equation 2:

$$\text{NDVI} = \frac{(R_{\text{NIR}} - R_{\text{Red}})}{(R_{\text{NIR}} + R_{\text{Red}})} \quad \text{Equation 2}$$

where R_{NIR} and R_{Red} represent surface reflectance for TM band 4 (760-900 nm) and band 3 (630-690 nm). NDVI is one of the most successful and well-known satellite VIs for identifying vegetated areas and determining vegetation cover and status (Carlson and Ripley 1997), and a well-established relationship between aboveground vegetation biomass and NDVI has been demonstrated in various ecosystems including the Sierra Nevada (Carlson and Ripley 1997; Cuevas-González et al. 2009; Fernández-Manso et al. 2014; Hansen and Schjoerring 2003; Ji et al. 2012; Myneni et al. 2001; Soenen et al. 2010; Zheng et al. 2004). For each burned pixel, an NDVI annual time series was constructed using the five pre-fire years (Years -5 through -1), the year of the fire (immediate post-fire, Year +0), and five post-fire years (Years +1 through +5).

To determine which environmental variables were significantly correlated with post-fire vegetation regrowth in mixed-conifer and red fir forests, Ordinary Least Squares (OLS) regression with spatial filtering was used to model post-fire NDVI values in Year +5 in mixed conifer and red fir forests. Topographic elements (elevation, slope, and aspect), climate variables (derived from PRISM), fire severity from MTBS, the year when fire occurred (hereafter 'disturbance year'), and post-fire NDVI in Year+1 were used as predictor variables in OLS regression. Climate variables were calculated as anomalies from 1972-2011 means. Post-fire climate anomalies in Year+0, Year+1, and the average anomaly from year +0 to year + 4 were used as predictor variables to explain the variation in post-fire NDVI in Year+5. Due to the large number of qualified burned pixels (Figure 21), random sampling was used. Between 30% and 40% of burned pixels were randomly selected without replacement to represent the population of the dependent variable (Year +5 NDVI) for red fir and mixed conifer forests.

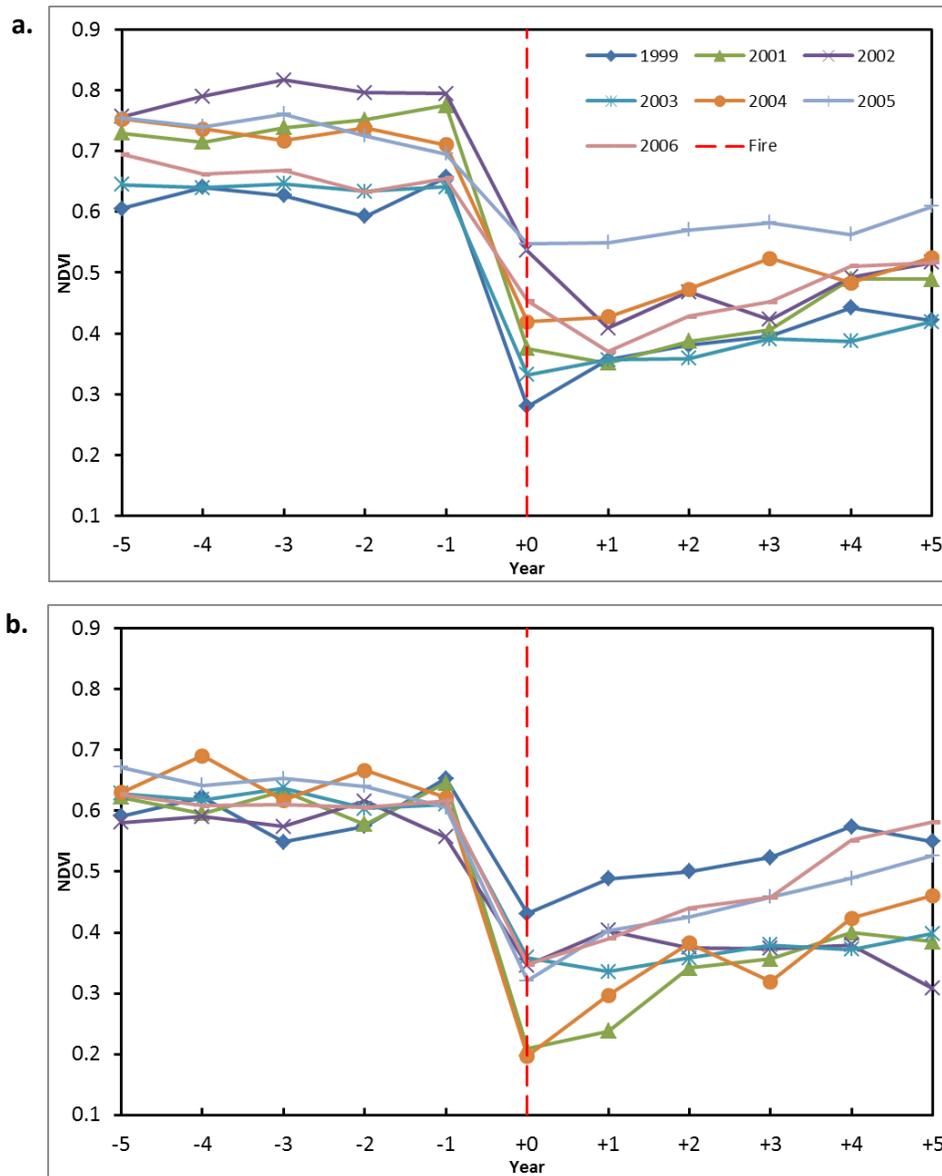
3.2.3 Results

As expected, there were abrupt drops in mean NDVI in Year +0 immediately following fire (Figure 22). In the post-fire period, the NDVI series of both forest types showed a general trend of progressively increasing mean NDVI as succession ensued; in the pre-fire period, the NDVI series showed consistent values with low temporal variability, indicating the stability in the forest cover until the occurrence of fire. The NDVI series of both forest types also indicated a higher temporal variability in the post-fire period than in the pre-fire period for most of disturbance years.

Although burned areas were generally characterized by increasing NDVI in years +0 through +5, the mean NDVI time series indicated pronounced differences in vegetation dynamics with forest types and fire years. Red fir NDVI increased in Year +1 in most of fire years examined, while in some years (fire years 2001, 2002 and 2006) mixed-conifer burned areas continued to regress and NDVI further decreased. All the mean NDVI series indicated a general trend towards pre-fire values over time following fires, but the recovery magnitude in NDVI depended on year in which fire occurred. Changes in NDVI from Year+1 to Year+5 were

spatially clustered (Figure 20), and also tended to be lower in red fir forests than in mixed conifer forests.

Figure 22: Time Series Mean NDVI Values, by Disturbance Year



a) mixed conifer forest b) red fir forest

To describe the global relationship between post-fire NDVI in Year+5 and environmental variables in mixed conifer and red fir forests, the multiple regressions with spatial filtering in equation 3 and equation 4 were adopted respectively, after dropping predictor variables with low significance frequency. In the following equation, β_0 is the intercept and β_1 to β_6 are the coefficients; $\beta_{Disturbance\ year}$ is the dummy variable (Year 1999, Year 2001 to Year 2006), and ε is the residual:

$$\text{Post-fire mixed conifer NDVI in Year +5} = \beta_0 + \beta_1 \text{Post-fire NDVI in Year +1} + \beta_2 \text{Slope} + \beta_3 \text{Northness} + \beta_4 \text{Post-fire wet season precipitation anomaly in Year +0} + \beta_5 \text{Post-fire}$$

January minimum temperature anomaly in Year+0 + β_6 Fire severity + $\beta_{Disturbance\ year}$ + ϵ
Equation 3

Post-fire red fir NDVI in Year + 5 = β_0 + β_1 Post-fire NDVI in Year +1 + β_2 Slope + β_3
Eastness + β_4 Elevation + β_5 Post-fire January minimum temperature anomaly in Year +0
+ β_6 Fire severity + $\beta_{Disturbance\ year}$ + ϵ Equation 4

Strong relationships were found between post-fire NDVI in Year+5 and five predictor variables (Post-fire NDVI in Year +1, slope, disturbance year, fire severity, and January minimum temperature anomaly in Year +0) for both mixed-conifer and red fir forests (Table 6, Table 7). However, northness and wet season precipitation anomaly in Year +0 were only robust for explaining variations in post-fire NDVI in Year+5 of mixed conifer forests (Table 6), and eastness and elevation were only robust for explaining variations in post-fire NDVI in Year+5 of red fir forests (Table 7). With regards to the role of environmental factors, several lines of evidence attested to the strong relationships between NDVI in Year +5 and predictor variables. The coefficient values for the predictor variables revealed a positive relationship between post-fire NDVI in Year +5 and two of the factors, slope and post-fire NDVI in Year+1, for both red fir and mixed-conifer forests; by contrast, a negative relationship existed between post-fire NDVI in Year +5 and fire severity. There was a positive relationship between post-fire NDVI in Year+5 and January minimum temperature anomaly in Year+0 in red fir forests, while the opposite trend existed in mixed-conifer forests. In addition, positive wet season precipitation anomalies in Year +0 predicted greater NDVI values in Year +5 in mixed conifer forests, while negative anomalies predicted lower NDVI values. One of the aspect-related variables—northness—was negatively related with post-fire NDVI in mixed-conifer, and another aspect-related variable—eastness—was positively related with post-fire NDVI in red fir forests. Also, at higher elevation, smaller post-fire NDVI values in Year +5 were found for red fir forests.

Table 6: OLS Modeling Results of Mixed-Conifer Forests with Spatial Filtering at the 95% Significance Level ^a

Variable	Coefficient	Std.error
Intercept	4280.951	108.147
Fire severity	-0.999	0.079
Post-fire NDVI in Year+1	0.093	0.014
Slope	12.033	2.416
Northness	-167.234	30.521
Post-fire January minimum temperature anomaly in Year +0	-4.84	0.655
Post-fire wet season precipitation anomaly in Year +0	158.180	16.591
Disturbance Year (dummy variable) ^b	[-166.798, 1671.782]	[30.322, 493.831]

^a Coefficients values based on scaled NDVI by 10,000; ^b Dummy variable related modeling results are provided in the form of value range; Residual standard error = 651.93; Multiple R-squared = 0.695; Adjusted R-squared = 0.675.

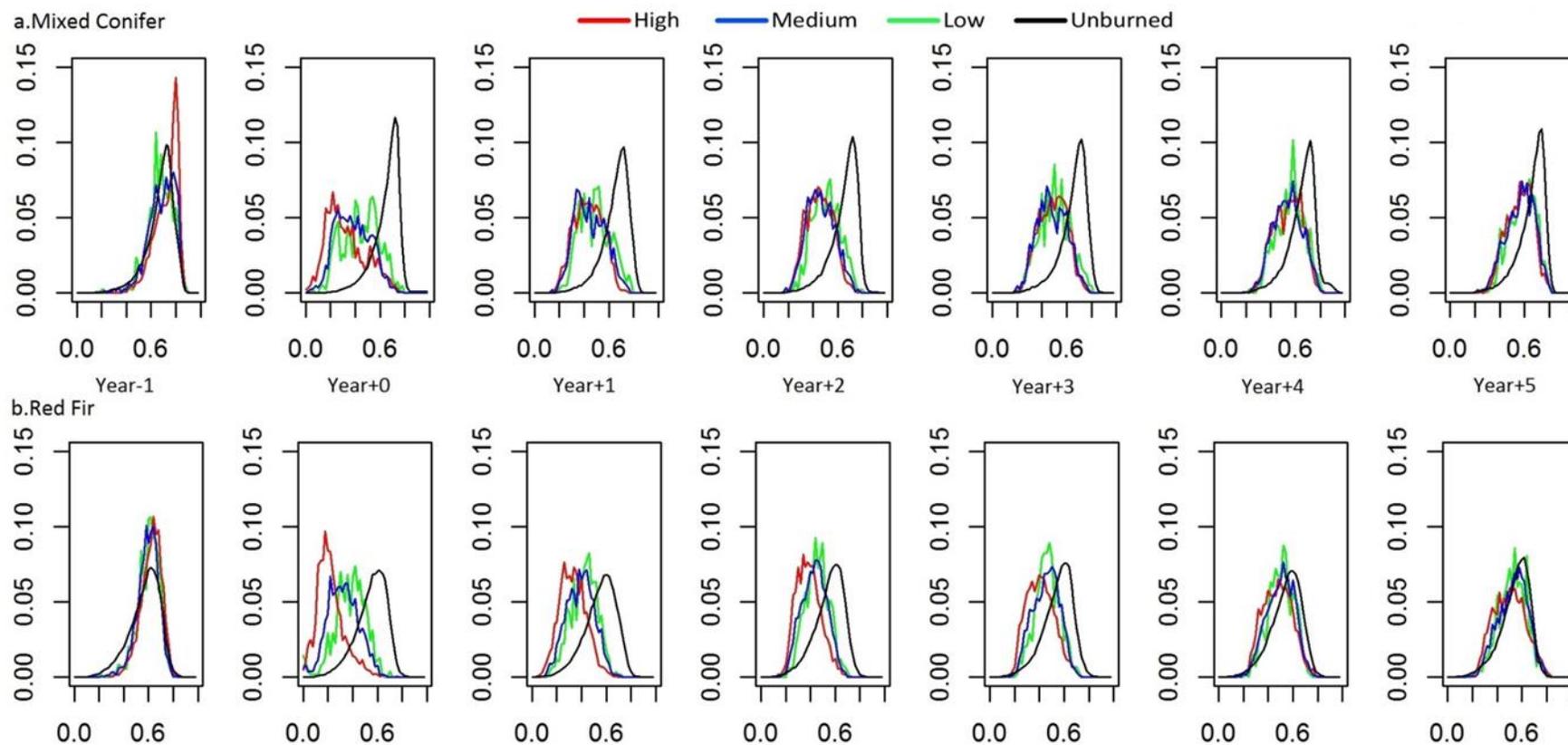
Table 7: OLS Modeling Results of Red Fir Forests with Spatial Filtering at the 95% Significance Level ^a

Variable	Coefficient	Std.error
Intercept	7514.30	399.726
Fire severity	-0.445	0.075
Post-fire NDVI in Year+1	0.198	0.017
Slope	14.587	2.820
Eastness	146.114	28.544
Elevation	-1.423	0.110
Post-fire January minimum temperature anomaly in Year +0	177.750	45.735
Disturbance Year (dummy variable) ^b	[-2079.174, 1830.554]	[294.206, 371.367]

^a Coefficients values based on scaled NDVI by 10,000; ^b Dummy variable related modeling results are provided in the form of value range; Residual standard error = 660.307; Multiple R-squared = 0.701; Adjusted R-squared = 0.684.

Time-series frequency densities of NDVI values were examined by fire severity for mixed-conifer and red fir forests (Figure 23). Abrupt changes in distributions can be observed in starting in Year +0 following fire for both forest types. Burned pixels didn't recover to pre-fire distributions, although differences between unburned and burned distributions decreased over time. NDVI values in high severity group were also both lower than in low-moderate groups in Year+0. However, apparent differences in the changing fire severity effects, especially between high and low-moderate groups, existed between mixed-conifer and red fir forests. Fire severity effects in mixed-conifer forests were apparent in Year+0, but disappeared quickly starting from Year+1; in contrast, apparent fire severity effects in red fir forests persisted for a longer time from Year+0 to Year+5, although they also diminished over time (Figure 23).

Figure 23: Frequency Distributions of NDVI Values by Fire Severity Class for Mixed conifer forest and Red Fir Forest



3.2.4 Discussion

Wildfire can leave patches with heterogeneous fire severity across the landscape, and fire severity is one of most important factors influencing post-fire vegetation succession change within patches (Collins and Roller 2013; Crotteau et al. 2013). In mixed-conifer and red fir forests of Sierra Nevada, the analysis found that the effects of fire severity on post-fire NDVI were still notable five years following fires (Figure 23). Lower NDVI values were expected in high severity burned areas in the early post-fire period. We can see that forest canopy recovered quickly in the low-moderate burned areas, but more slowly in the high burned areas. This phenomenon is consistent with previous studies that mixed-conifer and red fir forests could withstand low-moderate severity fire, but could not survive after the high severity fire (Scholl and Taylor 2006). In terms of NDVI values, differences in burn effects between high and low-moderate fire severity patches might be more notable and persistent in red fir forests at higher elevation than in mixed conifer forests at lower elevation (Figure 23). This may in part be due to the longer winter and shorter growing season in higher elevation red fir forests (Laacke and Tappeiner 1996).

Researchers tested and compared the sensitivity of post-fire vegetation dynamics in mixed conifer and red fir forests to precipitation and temperature. The significant relationships between the post-fire NDVI and climate anomaly variables indicate that post-fire climate in the first growing season might be one of the most important factors for forest succession in mixed conifer and red fir forests in the Sierra Nevada Mountains. In terms of precipitation, higher post-fire NDVI values were expected in mixed conifer forests with higher wet season precipitation, but this relationship did not exist in red fir forests. This corresponds to the general spatial pattern that water stress decreased with elevation at lower montane forests of the Sierra Nevada, and post-fire drought, defined as lower than normal precipitation and higher than normal temperature in this study, was unfavorable for tree regeneration after disturbances in Mediterranean environments (Broncano and Retana 2004; Daskalakou and Thanos 1996; Sánchez - Gómez et al. 2006).

In terms of temperature, January minimum temperature was significant for explaining post-fire NDVI values in both mixed conifer and red fir forests. The negative temperature relationship for mixed conifer forests could be a proxy for drought effects, but positive relationship for red fir forests could be a proxy for solar-radiation effects. The modeling results between post-fire vegetation cover and climate anomaly variables might be explained by the fact that in the Sierra Nevada Mountains, the limiting factor for vegetation growth at lower elevation is soil–water balance, and at higher elevation is solar-radiation (temperature) (Greenberg et al. 2009; Miller and Urban 1999b). An inverse relationship between elevation and post-fire NDVI in Year+5 in red fir forests may be attributable to decreasing temperature with increasing elevation in upper montane forests. These findings together suggest that, in this study area, cover reestablishment in mixed conifer forests at lower elevation was more sensitive to post-fire drought, but in red fir forests at higher elevation cover reestablishment was more sensitive to post-fire warmth.

Landscape position can also strongly influence post-fire vegetation regrowth through its effects on local microclimate and hydrological processes (Ireland and Petropoulos 2015; Mitchell and Yuan 2010). Regression models in the current study showed that north-facing aspects in mixed-conifer forests tended to have higher post-fire vegetation cover compared to south-facing exposures. This is consistent with studies (within the northern hemisphere) showing that south-facing aspects were

more prone to drought, and therefore regeneration appeared to be slower compared to north-facing aspects (Ireland and Petropoulos 2015; Mitchell and Yuan 2010; Wittenberg et al. 2007). In red fir forests, east-facing aspects exhibited significantly higher post-fire vegetation cover compared to west facing exposures.

3.3 Sierra Nevada Shrub Versus Tree Response to Seasonal Water

Availability

3.3.1 Rationale

There is widespread concern that changes in climate and fire regime may lead to vegetation change across California, which in turn may influence watershed hydrology. Although plant cover is known to affect numerous hydrological processes, sensitivities to vegetation type and spatial arrangement of species within watersheds are not well understood. To generate mechanistically-based projections of how potential type conversion from forested to shrub dominated systems may affect streamflow, these specific research questions were addressed:

- 1) During the peak growing season (early spring – fall), how does plant water availability differ between the dominant trees and shrubs at the snow-rain transition zone in the Sierra Nevada?
- 2) How does variation in water availability between trees and shrubs manifest in leaf-level gas exchange rates, namely stomatal conductance (the primary control on photosynthesis and transpiration rates)?

Assuming that trees have a more extensive rooting distribution than shrubs (Canadall et al. 1996), one can hypothesize that conifer trees will maintain higher leaf-gas exchange rates during the dry season than more shallowly-rooted shrubs. Alternatively, while trees may have greater access to stable water resources during the dry season, they also need to support higher leaf area and thus may implement a more conservative water use strategy relative to shrubs (Mencuccini, 2003; Veneklaas and Poot 2003).

This study is one the few to compare the physiology of shrubs versus trees in this ecosystem (DeLucia and Schlesinger, 1991; Royce and Barbour 2001). A key outcome of the study was developing a relationship between plant water availability and use, which was then used to improve estimates of modeled streamflow from RHESSys, a process-based eco-hydrological model. This research demonstrates how linking strategic field data collection and mechanistic ecohydrologic models can be used as a robust tool for assessing the potential impact of vegetation change on the water balance of an ecosystem. This is an increasingly valuable approach for informing management decisions focused on adapting strategies based on projected changes in climate.

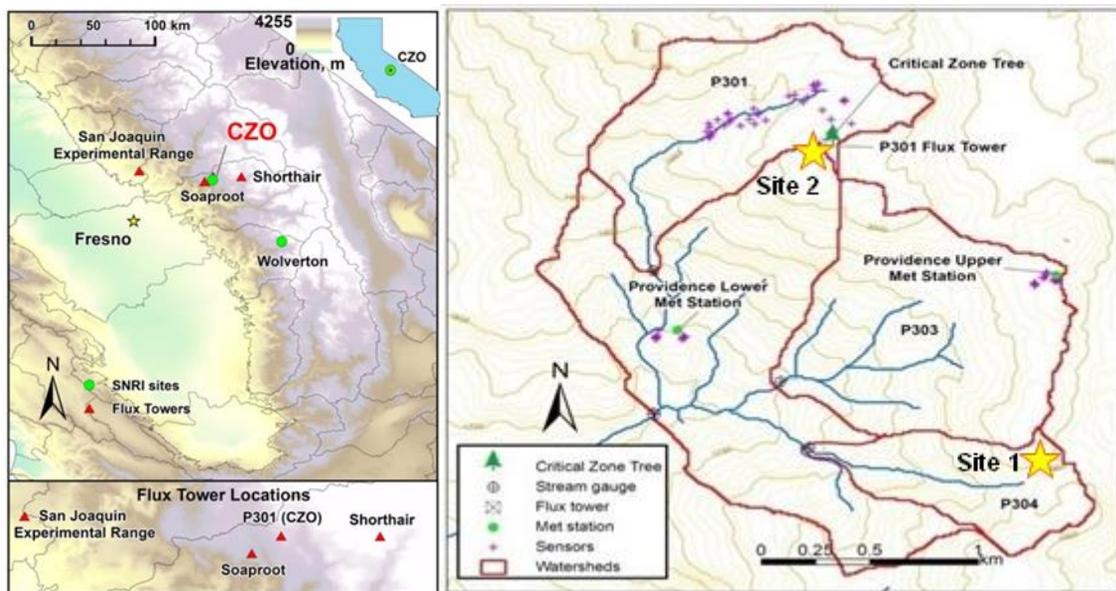
3.3.2 Sierra Nevada Water Use Site Location and Species Selection

The investigators selected field sites within a mixed conifer forest at the snow-rain transition zone in the California Sierra Nevada (approximate elevation range: 1950-2000 m). The field sites were located within the Sierra Nevada Critical Zone Observatory (SSCZO) Providence Creek watershed,

which is divided into sub-watersheds (referred to as P301 through P304) (Figure 24). These watersheds are used by the U.S. Forest Service (USFS) and researchers across California to improve our understanding of the geologic, climatic, and biotic controls on hydrologic function in these mountains. Within the SSCZO watershed, two sites were selected that consisted of large patches of dominant montane chaparral species in an otherwise conifer dominated landscape (Figure 24). The focal species in the study consisted of two dominant conifer species: White fir (*Abies concolor*) and Incense cedar (*Calocedrus decurrens*), and two dominant shrub species: Greenleaf manzanita (*Arctostaphylos patula*) and Mountain whitehorn (*Ceanothus cordulatus*). Five mature individuals per species were selected from each site (i.e., n=5 per species per site).

Site 1 is a relatively flat, open area at 1930 m near the top of the P304 watershed (37.051 N, 119.184 W). Soils at this site were deep and relatively sandy, derived from granodiorite rock. The relative dominance of shrublands at this site appears to be at least partially related to disturbance, e.g., logging, as various tree species are regenerating in the area. Climax vegetation is likely a mixed conifer forest of White fir, Incense cedar (*Calocedrus decurrens*), and pine (*Pinus lambertiana*, *P. jeffreyi*). To observe changes in microclimate conditions during the field season, the team instrumented Site 1 with temperature (C°) and relative humidity (%) sensors (model U23 Temperature/Relative Humidity dataloggers, Onset HOBO Pro v2, Onset Computer Corp. Bourne, MA) at five locations. Microclimate data was recorded every 30 minutes between June-Oct. 2014.

Figure 24: Southern Sierra CZO Instrumentation at Providence Creek



Maps include meteorological and stream instrumentation from the Kings River Experimental Watershed (a USDA Forest Service project) (left). Field sites within the Providence Creek watershed are indicated by yellow stars (right).

Source: Map credit: Matt Meadows, <http://criticalzone.org/sierra/infrastructure/field-area/providence-creek-headwaters/#overviewMaps>

Site 2 is at slightly higher elevation (2100 m) in the P301 sub-watershed. Soil depth is highly variable at this site, but a recent study has shown that soils are highly porous near the surface and that saprolite is a crucial water reservoir at depth (Holbrook et al. 2014). Shrub patches at this site were more spatially heterogeneous than at Site 1. Site 2 is also the location of the SSCZO eddy covariance tower, a highly instrumented tower that measures ecosystem-scale carbon dioxide and water vapor fluxes (Goulden et al. 2012). The focal study plants were within the footprint of the flux tower.

3.3.3 Methods

3.3.3.1 *Quantifying seasonal variation in plant water availability and use*

Within each site, species replicates were grouped and stratified by microhabitat conditions. During the 2014 peak growing season (April – October), field crews measured the ecophysiological responses (plant water potential and leaf gas exchange rates) of the two dominant tree and shrub species to changes in seasonal water availability at the two field sites within the SSCZO.

A commonly used and widely accepted metric of plant water availability at the rooting zone is predawn leaf water potential (Ψ_{pd} , MPa). During predawn hours (0200-0330 hr), plants are assumed to be at equilibrium with soil water potential at their rooting zone; therefore, this is when plants are in their most hydrated state (Lambers et al. 2008). Ψ_{pd} values closer to zero indicate that plants are more hydrated than if values are larger and more negative. Crews measured Ψ_{pd} every three weeks between 6 April and 24 October 2014 from tree and shrub species at each of the two sites using a Scholander pressure chamber (Model 1000, PMS Instruments Inc., Albany, OR). They collected two twig samples per individual per species at each site.

To assess how carbon gain (photosynthesis) and leaf water loss (transpiration) varied between functional groups during the growing season, they collected measurements of maximum leaf-level gas exchange rates per unit leaf area using a portable gas-exchange system (LI-COR 6400XT, LI-COR Biosciences, Lincoln, NE). A specific project goal was to relate maximum stomatal conductance, i.e., the degree to which stomata are open or closed, to Ψ_{pd} . The relationship between these response variables would provide insight into how trees versus shrubs regulate water use as residual soil water declines during seasonal dry down (Reich and Hinckley, 1989).

To isolate the effects of leaf-level responses to changes in plant water availability, researchers set the environmental conditions within the leaf cuvette to simulate the environmental conditions at which these species typically photosynthesize at maximum rates (Running, 1976; Mission et al., 2006). Photosynthetically active radiation (PAR) was set to 1800 nm, leaf temperature was 25 °C, and concentration of carbon dioxide was set to 400 ppm. Similar environmental parameters were used in a comparable study within a mixed-conifer forest in the Sierra Nevada (Mission et al. 2006).

3.3.3.2 *Data Analysis*

For these analyses, the team pooled species by functional types (trees: White fir and Incense cedar; shrubs: Greenleaf manzanita and Mountain whitehorn). For Ψ_{pd} measurements, they calculated the average Ψ_{pd} of both twig samples per individual from all species at each site. They then calculated the average Ψ_{pd} value per species prior to pooling by functional types. Each functional group was also pooled across sites because site was not a significant predictor of the physiological response within functional groups. Pooling across site increased the sample size for each functional

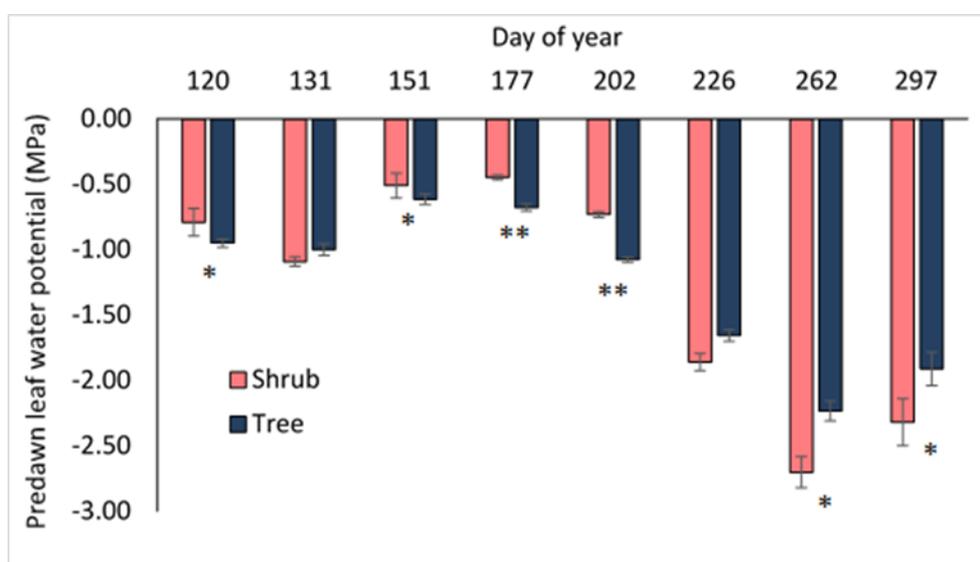
group (n=20 individuals per species). A repeated measures ANOVA was performed to test for statistical differences in ecophysiological responses between tree and shrub species over time. If significant ($P < 0.05$), investigators performed a pairwise comparison to identify points in time when these functional groups differed significantly from one another. Statistical analyses were performed using JMP version 10.0.0 software (SAS Institute, Cary, SC).

To assess the strength of the relationship between plant water availability and use for shrubs and trees, they first normalized average stomatal conductance for each sampling period by the seasonal maximum and plotted these values against average predawn leaf water potential. Analysis of the best fit relationship to these data was performed using R-Studio version 2.12.2 (R development Core Team 2012).

3.3.4 Results

Plant available water was not consistently different between trees and shrubs through the growing season (Figure 25). Shrubs and trees had similar access to water through the early part of the growing season (April-early June) but began to diverge in late-June (day of year, DOY=177). In late-June, available water to shrubs was twice that of trees (Figure 25; shrubs: $\Psi_{pd} = -0.55 \pm 0.08$ MPa; trees: $\Psi_{pd} = -1.07 \pm 0.08$ MPa, $P < 0.05$). This trend changed at the peak point of the dry season (mid-September, DOY=262) when trees were more hydrated than shrubs (Figure 25; shrubs: $\Psi_{pd} = -2.70 \pm 0.80$ MPa; trees: $\Psi_{pd} = -2.23 \pm 0.60$ MPa, $P < 0.05$), and this pattern lasted until early autumn (DOY=297).

Figure 25: Average Predawn Leaf Water Potential (MPa) for Trees and Shrubs During the Growing Season



Shrubs had higher leaf-level gas exchange rates per unit leaf area than trees through the growing season (Figure 26). Stomatal conductance (g_s), which correlates strongly with photosynthetic rates ($r^2=0.70$, $P=0.0001$), was twice as high for shrubs than trees in April (DOY=120), which is when snow was observed beginning to melt (Figure 26; shrub g_s : 0.11 ± 0.02 mol H_2O $m^{-2} s^{-1}$; tree g_s : 0.05 ± 0.009 mol H_2O $m^{-2} s^{-1}$). In late-June (DOY=177), the difference in stomatal conductance was

magnified between shrubs and trees (Figure 26; shrub g_s : $0.40 \pm 0.06 \text{ mol H}_2\text{O m}^{-2} \text{ s}^{-1}$; tree g_s : $0.09 \pm 0.007 \text{ mol H}_2\text{O m}^{-2} \text{ s}^{-1}$), and this difference persisted through the end of the peak growing season in September and October (Figure 26).

The difference in g_s was manifested in carbon assimilation rates (Table 8). Maximum photosynthetic rates (A_{max}) in shrubs were approximately three times that of trees in July (Table 8; shrub A_{max} = $21.4 \pm 1.4 \mu\text{mol m}^{-2} \text{ s}^{-1}$; tree A_{max} = $7.9 \pm 0.7 \mu\text{mol m}^{-2} \text{ s}^{-1}$). The lowest photosynthetic rates were observed in late-October for both trees and shrubs (Table 8).

Figure 26: Differences in Stomatal Conductance (g_s , $\text{mol m}^{-2} \text{ s}^{-1}$) Between Trees and Shrubs During Growing Season

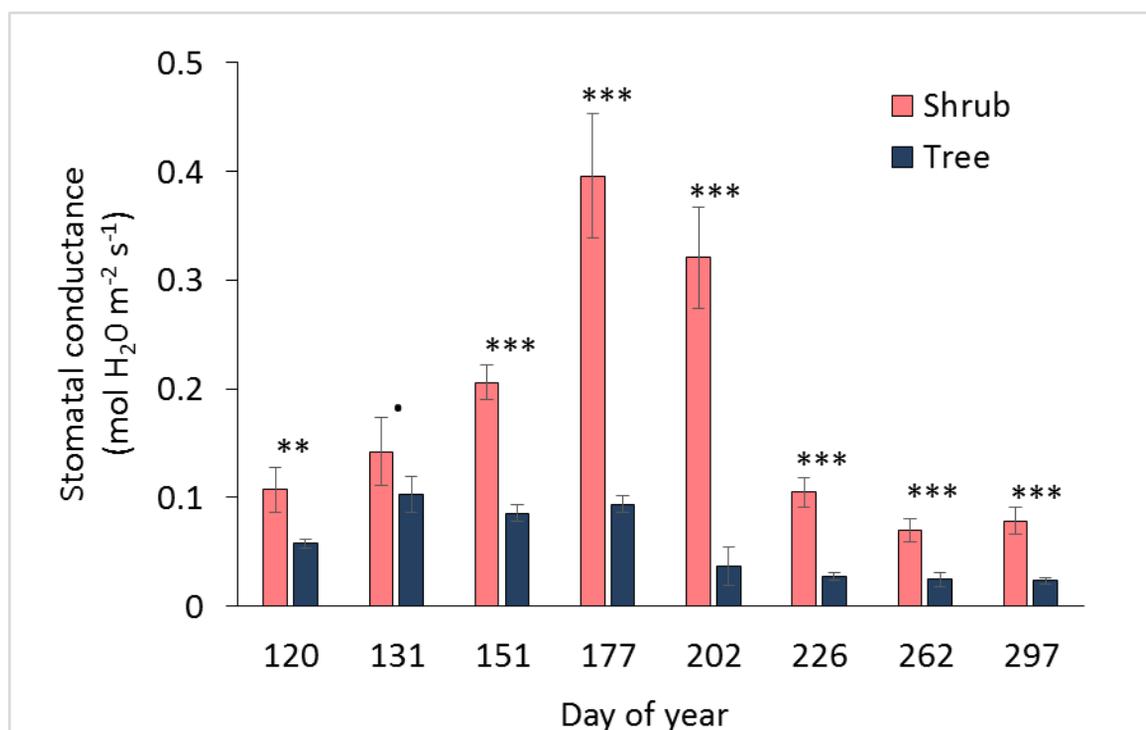
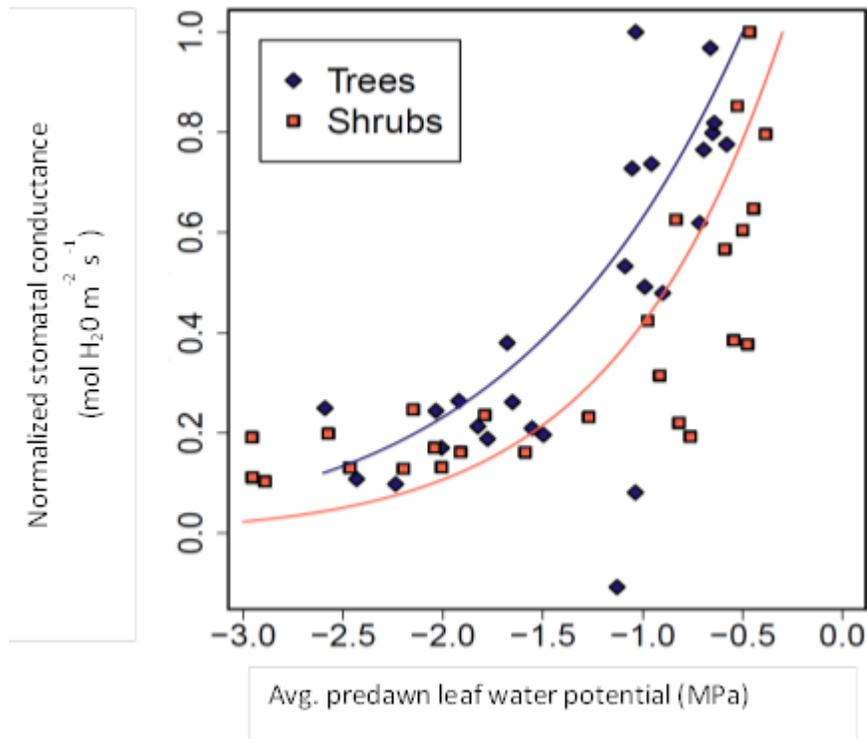


Table 8: Seasonal variation in average stomatal conductance (g_s , mol m⁻² s⁻¹) and maximum photosynthetic rates (A_{max} , μ mol m⁻² s⁻¹) in trees and shrubs during the growing season

DATE		30-APR	11-MAY	31-MAY	26-JUN	21-JUL	14-AUG	19-SEP	24-OCT
	Day of year	120	131	151	177	202	226	262	297
SHRUB	g_s	0.11 ± 0.02	0.14 ± 0.03	0.21 ± 0.02	0.40 ± 0.06	0.32 ± 0.05	0.10 ± 0.01	0.07 ± 0.01	0.08 ± 0.01
	A_{max}	9.5 ± 1.1	9.9 ± 1.4	15.3 ± 0.9	21.4 ± 1.8	20.7 ± 1.6	10.6 ± 1.1	7.9 ± 1.2	6.5 ± 1.3
TREE	g_s	0.06 ± 0.001	0.10 ± 0.02	0.09 ± 0.01	0.09 ± 0.01	0.04 ± 0.02	0.03 ± 0.001	0.02 ± 0.01	0.02 ± 0.001
	A_{max}	7.0 ± 0.4	9.5 ± 1.7	5.8 ± 0.7	7.9 ± 0.7	8.8 ± 1.5	4.1 ± 0.8	5.6 ± 1.7	3.2 ± 1.3

Despite the consistently higher rates of g_s in shrubs relative to trees during the growing season (Figure 26), shrubs and trees had similar sensitivity to changes in water availability (Figure 27). Even at minimum (most negative) values of Ψ_{pd} , both shrubs and trees kept stomata open ($g_s > 0 \text{ mol m}^{-2} \text{ s}^{-1}$) (Figure 27; shrubs: $\Psi_{pd} = -2.6 \text{ MPa}$, norm. $g_s = 0.2 \text{ mol m}^{-2} \text{ s}^{-1}$; trees: $\Psi_{pd} = -2.0 \text{ MPa}$, norm. $g_s = 0.26 \text{ mol m}^{-2} \text{ s}^{-1}$).

Figure 27: Relationship Between Changes Predawn Leaf Water Potential (MPa) and Stomatal Conductance (g_s , $\text{mol m}^{-2} \text{ s}^{-1}$) Normalized by Maximum Rates of g_s for the Season.



3.3.5 Discussion

These results reveal that shrubs and trees have similar regulatory mechanisms of water use as soil dries. The study found that g_s declined sharply between predawn leaf water potential values of -0.5 and -1.25 MPa for both plant functional types (Figure 27). For the suite of species in this study, these values of Ψ_{pd} when a steep decline was observed in g_s are significantly more positive than reported thresholds of Ψ_{pd} beyond which hydraulic failure is likely to occur (Choat et al. 2012). Therefore, these results strongly suggest that trees and shrubs have conservative water use strategies as antecedent soil is depleted. Moreover, stomatal regulation appears to be a key mechanism for maintaining water status during the dry season for both plant functional types.

While the sensitivity to changes in water availability during the dry season may be similar between the dominant trees and shrubs in the Providence Creek watershed, shrubs maintained much higher rates of leaf gas-exchange per unit leaf area at lower (more negative) Ψ_{pd} values than trees. DeLucia and Schlesinger (1991) found similar patterns between Sierran tree and

shrub species: shrub species had lower water-use efficiency (carbon gain per unit water loss) than tree species, and this was thought to be advantageous to shrubs in a competitive, water-limited environment. The results in the present study are also in accordance with findings by Royce and Barbour (2001), who observed the same shrub species in the Kern plateau of southern Sierra Nevada depleted soil moisture faster than conifer trees did. Unlike Royce and Barbour (2001), this study rarely observed Ψ_{pd} values become more negative than -2.5 MPa for either trees or shrubs, even in September when Ψ_{pd} values were at their minimum values. Moreover, the difference in Ψ_{pd} values between trees and shrubs was much smaller through the growing season than observed by Royce and Barbour (2001), who found that Ψ_{pd} values in shrubs were at least -2 MPa more negative than trees by the end of summer.

Taken together, the present results support the idea that shrubs possess leaf traits and/or life history characteristics that allow higher rates of carbon fixation even as soil water becomes depleted relative to trees. One strategy may be related to leaf hydraulic traits (Bartlett et al. 2012). It is reasonable to hypothesize that shrubs adjust the concentration of solutes in their cells to support greater leaf hydration as soil dries (Bowen and Roberts 1985). Another possible explanation for higher transpiration rates of shrubs than trees could be phenological differences that impact plant water requirements and/or water-use efficiency (Field and Mooney, 1983). The study observed both shrub species at each site dropped most of their older leaves in July and put on new leaves in August, which likely reduced the water requirement of shrubs temporarily as the total leaf area declined (Medina and Francisco 1994).

The patterns observed in leaf-gas exchange rates between trees and shrubs were contrary to expectations. The authors assumed that trees, which are expected to have a more extensive rooting distribution than shrubs, would maintain higher leaf-gas exchange rates during the dry season than more shallowly rooted shrubs. Instead, results lend greater support to an alternative hypothesis: while trees may have more access to water, they need to support higher leaf area, thus utilizing conservative water use strategies relative to shrubs. Overall, a less conservative water use strategy exhibited by shrubs may make shrubs better competitors than trees for limited water resources in a future climate expected to become warmer, and possibly drier.

CHAPTER 4:

Vegetation Type Conversion: Effects on Stream Flow

4.1 Rationale

Vegetation type conversion is predicted to be a major source of land-cover change in California due to the combined effects of climate change and vegetation disturbances (e.g., fire, disease). In the dry mountainous regions of the Western US, rising temperatures increase evaporative demands on forests, making them more vulnerable to dieback (Williams et al., 2013) and increasing the likelihood of type conversion to shrubland. In the coastal watersheds of southern California, increasing fire frequencies have the potential to transform shrubland-dominated landscapes to grass-dominated landscapes (Zedler et al., 1983).

In both of these scenarios, numerous hydrological processes may be affected by vegetation type-conversion. First, reductions in above-ground biomass decrease rainfall interception, increasing the amount of water at the soil surface available for infiltration or runoff. Second, changes in vegetation properties such as LAI, rooting depth, and maximum stomatal conductance may affect transpiration levels. Higher temperatures associated with climate change may also alter snowpack dynamics in mountainous regions. These processes, in turn, may affect different aspects of the streamflow regime, including total yield, peak flows, low flows, and the timing of flows.

In southern California, the effect of type conversion from chaparral shrublands to grassland on streamflow remains uncertain. While small-scale conversion experiments from chaparral to grass in studies from southern California have indicated that streamflow may increase (Hill and Rice, 1963) and Arizona (Hibbert, 1971), no studies have evaluated the effect of type conversion along with climate change on streamflow.

In the Sierra Nevada, many studies have evaluated the effects of climate change on streamflow (Null et al., 2010; Young et al., 2009) and hydropower (Madani and Lund, 2010; Rheinheimer et al., 2014). However, all of these studies were conducted under the assumption that vegetation is stationary. Recently, Goulden and Bales (2014) have suggested that changes in transpiration associated with vegetation change in the Sierra Nevada may significantly alter streamflow totals. However, no known modeling studies have investigated the effect of type conversion, in combination with climate change, on streamflow in the Sierra Nevada.

Streamflow originating in the Sierra Nevada is a critical source of water supply for agriculture, urban and environmental uses, as well as a provider of hydropower generation. Hydropower in the Sierra represents an average of 16% of the total energy production in the state of California (Rheinheimer et al., 2014). Understanding how forest-to-shrub type-conversion may affect hydropower in California necessitates understanding how changes in streamflow may affect reservoir capacity in California. Numerous studies have previously investigated the effect of climate change on reservoir hydropower in the Sierra Nevada (Guegan et al., 2012; HRR-GWRI, 2011). This study compliments these previous studies by examining the effect of vegetation type-conversion on reservoir capacity for hydropower production.

In this chapter, an ecohydrologic modeling approach is used to investigate the effects of shrub-to-grass type conversion in coastal southern California and tree-to-shrub type conversion on streamflow in the Sierra Nevada. The effect of type conversion in the Sierra Nevada is then extended to include the associated impact on reservoir capacity.

4.2 Description of RHESSys Model

This study estimated watershed-scale hydrologic impacts of species change using RHESSys (Tague and Band, 2004). RHESSys has been continually updated since 2004. This project utilized version 5.19 (additional details and model code are available through <https://github.com/RHESSys/RHESSys/>). RHESSys fully couples sub-models of terrestrial ecosystem carbon and nitrogen cycling with a spatially distributed hydrologic model. Thus RHESSys can account for both the impact of water availability on ecosystem productivity and growth and conversely the impact of vegetation growth on soil water availability and streamflow. Vertical water fluxes represented in RHESSys include interception, transpiration, soil, litter and canopy evaporation, infiltration, surface detention evaporation, soil accumulation and melt and drainage through a rooting zone soil layer. Energy, wind, and water are attenuated by the aboveground canopy using standard approaches (Tague & Band, 2004). Lateral moisture fluxes in RHESSys can occur via surface water (generated as either infiltration or saturation excess flow), shallow subsurface flow between spatially explicit patches, or through recharge to a deeper groundwater store. The ecosystem carbon cycling model accounts for photosynthesis, respiration, and allocation of net photosynthate to plant components (leaves, stems and roots) as well as turnovers of these components. RHESSys includes a moderately complex vegetation water use model that accounts for the impact of changes in soil water availability, atmospheric drivers (radiation, vapor pressure deficit and temperature), and plant structure (LAI, height, rooting depth and species type) on transpiration losses as well as evaporation losses via interception of canopy, litter, and soil water. RHESSys is run at a daily time step.

RHESSys has been used in prior studies to assess land use change and climate change impacts on water resources, carbon and forest mortality risk in the western U.S. and elsewhere. These studies demonstrate good performance using RHESSys through comparison with streamflow (e.g., Tague et al. 2013a, Tague and Peng 2013), carbon flux tower data (e.g., Zierl et al. 2006), and tree-rings/mortality data (e.g., Tague et al., 2013b). RHESSys has previously been applied in study sites similar to those used in this project, including forested watersheds in the California Sierra (e.g., Tague and Peng, 2013) and in the chaparral-dominated watersheds of Southern California (Tague et al., 2008; Shields et al, 2014).

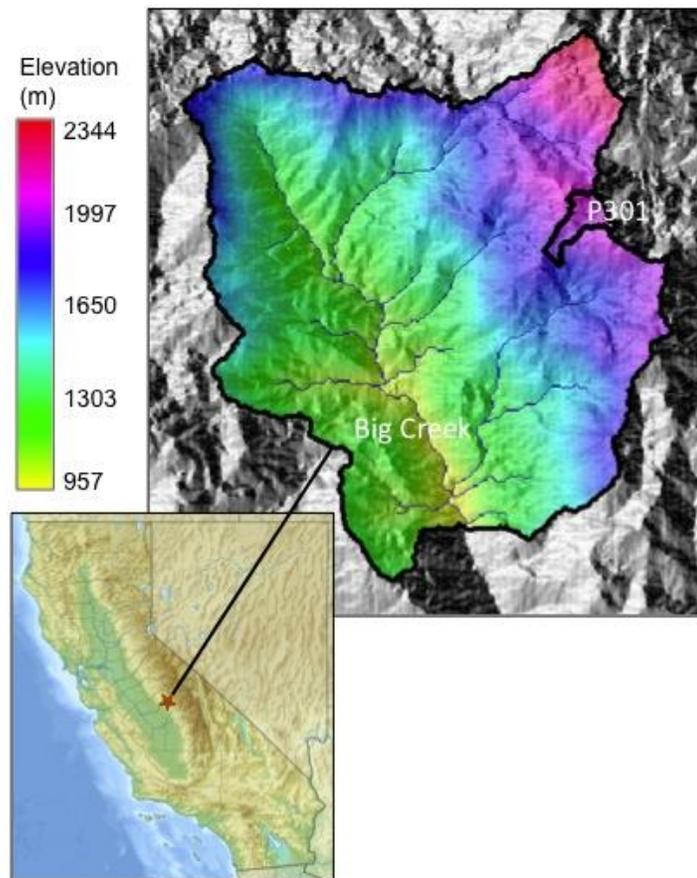
4.3 Parameterization of RHESSys for Chaparral & Sierran Vegetation Change

4.3.1 Watershed Descriptions and Data

Post-conversion streamflow change for the Sierra Nevada was investigated in the P301 and Big Creek watersheds located near Shaver Lake, California (Figure 28). Initial modeling was conducted in P301 and then scaled up to the larger Big Creek watershed to investigate the

effects of type conversion on reservoir operations (see Section 4.6 for description of Big Creek). P301 is a 99 ha watershed located at the head of Providence Creek within the Southern Sierra Critical Zone Observatory (SSCZO). P301 has an elevation ranging from 1790 to 2115 m and lies squarely in the rain-snow transition zone of the Sierra Nevada, with the snow fraction ranging from 35% to 60% of total precipitation (Hunsaker et al., 2012). Mean annual precipitation for P301 is approximately 1320 mm, while mean annual streamflow is 528mm (Hunsaker et al., 2012).

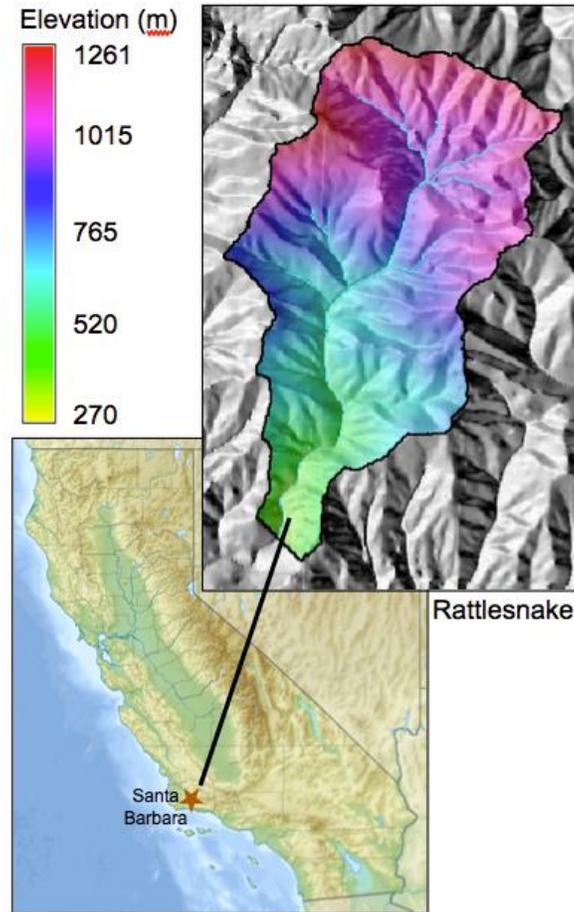
Figure 28: Sierra Nevada Watershed Locations.



The vegetation in P301 is primarily Sierran mixed-conifer forest with a small proportion of chaparral shrublands and meadows. Mixed conifer forest is composed of white fir, incense pine, ponderosa pine, Jeffery pine, and sugar pine. ET measurements from a flux tower located within the boundaries of P301 suggest that forest vegetation is active year-round, with little shutdown of ET during the winter or the dry summer (Goulden et al., 2012). The soils in P301, which are categorized as Gerle-Cagwin and are granitic in origin (Hunsaker et al., 2012), are generally very deep (Dahlgren et al., 1997).

The Rattlesnake Creek watershed was used to investigate chaparral-to-grass type conversion and is a sub-catchment of the larger Mission Creek watershed; a semi-arid, coastal catchment in Santa Barbara, California (Figure 29). The Rattlesnake watershed is approximately 6 km² and occupies the upper, undeveloped portion of the watershed where the terrain is steep. The watershed is dominated by the chaparral plant community and the Mediterranean climate here is characterized by warm, dry summers and cool, wet winters. Elevation ranges from 270m to 1261m in the Rattlesnake watershed.

Figure 29: Santa Barbara Watershed Location



The RHESSys model was set up for the Rattlesnake watershed following work by Shields and Tague (2012) in which Rattlesnake was modeled as a part of the larger Mission Creek watershed model. The stream gauge at Las Canoas Bridge (RS02) was used to define the watershed outlet. The model was run with a 10m DEM, and a patch spatial unit of 30m (the highest spatial resolution in RHESSys, where processes such as soil moisture and Carbon/Nitrogen cycling occur). The soil type was parameterized to characterize the steep slopes characteristic of Rattlesnake, and the vegetation was parameterized to reflect the chaparral plant community using literature values for chaparral species (White et al., 2000; Miller, 1981). Precipitation data

was obtained from a gauge near the Stanwood fire station (228), and the temperature data comes from a gauge located near the Santa Barbara harbor at 3m elevation. Precipitation was transformed using a temperature-elevation lapse rate to adjust the 3m data to 700m (Shields and Tague, 2012). From the precipitation and temperature records, meteorological data input for the model was constructed from 10/1/1988 through 6/7/2010. Streamflow data from the RS02 gauge from 12/6/2001 through 9/29/2011 was obtained from the Santa Barbara Coastal Long Term Ecological Research site.

4.3.2 Model Parameterization and Calibration

Field measurements of leaf water potential and stomatal conductance were used to improve parameterization of the RHESSys model. Stomatal conductance (gs) in the model is calculated based on a modified Jarvis multiplicative model (Jarvis, 1976); where leaf water potential (LWP), absorbed photosynthetically active radiation ($APAR$), CO_2 , vapor pressure deficit (VPD), and average and minimum temperature (t_{avg} and t_{min}) are normalized multipliers which scale maximum stomatal conductance (gs_{max});

$$gs = (m_{LWP} * m_{APAR} * m_{CO_2} * m_{VPD} * m_{t_{avg}} * m_{t_{min}}) * gs_{max}. \quad \text{Equation 5}$$

For both the Sierra Nevada and Santa Barbara, tree and shrub values of gs_{max} were derived from highest observed field measurements of stomatal conductance (averaged over samples from each species).

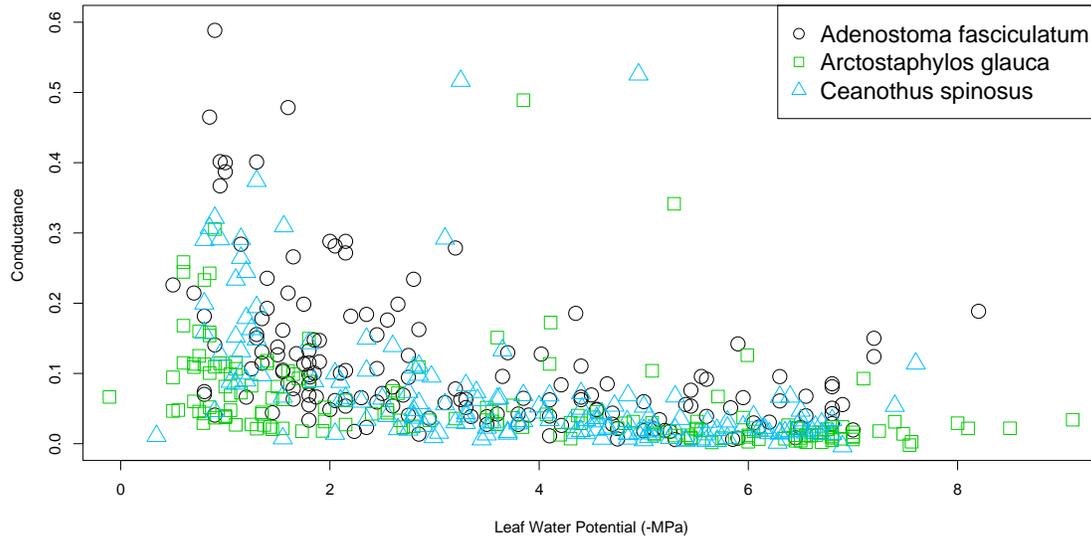
In the Sierra Nevada, a relationship between leaf water potential and the normalized LWP multiplier in Equation 5 was derived from the power law relation observed between predawn LWP and normalized average gs in Figure 27,

$$m_{LWP} = (slope * (LWP - LWP_{thresh}) + intercept)^p \quad \text{Equation 6}$$

where $slope$ was 0.1, $intercept$ was 1, LWP_{thresh} was -0.5 for trees and -0.3 for shrubs, and p was 9 for trees and 12 for shrubs. m_{LWP} has a minimum of 0 and a maximum of 1. This relationship was used to replace the prior linear submodel of LWP control on stomatal conductance.

For Rattlesnake, predawn leaf water potential and stomatal conductance were collected for three species found in the chaparral community: *Arctostaphylos glauca* (ARGL), *Adenostoma fasciculatus* (ADFA), and *Ceanothus spinosus* (CESP). Each of the three chaparral species showed differences in maximal stomatal conductance (Figure 30), however, the relation between leaf water potential and normalized stomatal conductance could not be separated statistically. For this study, the same linear LWP-conductance curve was used for all three species. Further, since streamflow estimates using the three different maximum stomatal conductances were nearly identical (less than 1% difference in annual flow), maximum stomatal conductance was set to the mean of observed maximum stomatal conductance for the three species.

Figure 30: Relation Between Stomatal Conductance and Leaf Water Potential



RHESys subsurface drainage parameters are typically calibrated to account for uncertainty in subsurface storage and drainage rates that are initialized based on soil texture (Tague et al., 2013a). RHESys was calibrated in both the Sierra Nevada and Santa Barbara using a Monte Carlo simulation to identify the optimal values of six parameters; lateral and vertical saturated hydraulic conductivity at the surface (K), lateral and vertical exponential decay of saturated hydraulic conductivity with depth (m), soil air entry pressure (pa), pore size index (po), percent of precipitation that bypasses the soil matrix to a deeper groundwater store (GW1), and percentage of deep groundwater store that moves to the stream (GW2).

Modeled streamflow was compared to the observed streamflow using a hybrid objective function, Accuracy, which is the product of Nash Sutcliffe Efficiency (NSE), NSE_{log} , and total percent error (PerErr);

$$NSE = 1 - \frac{\sum(Q_{obs} - Q_{mod})^2}{\sum(Q_{obs} - \overline{Q_{obs}})^2} \quad \text{Equation 7}$$

$$NSE_{log} = 1 - \frac{\sum(\log(Q_{obs}) - \log(Q_{mod}))^2}{\sum(\log(Q_{obs}) - \log(\overline{Q_{obs}}))^2} \quad \text{Equation 8}$$

$$PerErr = \frac{(\overline{Q_{mod}} - \overline{Q_{obs}})}{\overline{Q_{obs}}} \quad \text{Equation 9}$$

where Q_{obs} is observed streamflow and Q_{mod} is modeled streamflow. Values of NSE and NSE_{log} range from negative infinity to 1, however, only parameter sets with values above 0 were considered behavioral and included further in the study. Similarly, parameter sets with PerErr values less than -1 or greater than 1 were considered non-behavioral and excluded from the analysis. Accuracy for the model was then calculated as

$$Accuracy = NSE * NSE_{log} * (1 - |PerErr|). \quad \text{Equation 10}$$

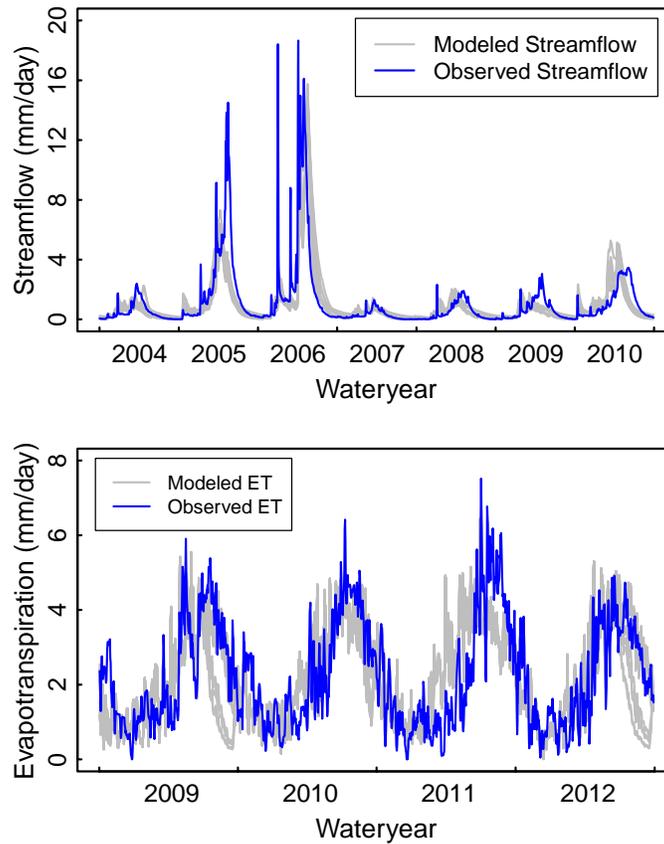
For the Sierra, RHESSys was calibrated against the observed streamflow record in P301 to identify the parameter sets that best represent the hydrologic processes in that watershed. Since ET data from the flux tower in P301 provided an additional measure in which to falsify parameter sets (Son and Sivapalan, 2007), RHESSys was also calibrated at the patch level against observed ET. The model was calibrated using seven years of streamflow data (wateryears 2004-2010) and four years of ET data (wateryears 2009-2012). For both calibrations, identical sets of 2000 parameter sets were sampled. Streamflow data recorded at the outlet of P301, as well as precipitation and temperature data measured at a meteorological station located just outside the watershed, were obtained from the US Forest Service Pacific Southwest Research Station. ET within the P301 was measured with a flux tower operated by the SSCZO. To account for parameter uncertainty during calibration, the ten parameter sets that gave the highest values of streamflow accuracy (Equation 10) *and* had ET accuracy values greater than zero were selected for simulation.

Results from P301 model calibration against observed streamflow and observed ET are provided in Table 9 and Figure 31. The model did a good job of replicating the behavior of both data sets, with streamflow being the better of the two. The best streamflow parameter sets topped out at 0.53 and 0.79 for NSE and NSE_{log} , respectively, while bias ranged from -6% to -22% (Table 9). Accuracy values for the top 10 streamflow parameter sets ranged from 0.3 to 0.36 for streamflow. Figure 31 shows the fit of the modeled streamflow and modeled ET compared to their respective observed values.

Table 9: Calibration Results for P301

Objective Function	Streamflow	ET
NSE daily	0.49 to 0.53	0.24 to 0.49
NSE_{log} daily	0.64 to 0.79	0.02 to 0.26
% BIAS (annual)	-6 to -22	-10 to 5
Accuracy	0.30 to 0.37	0.005 to 0.01

Figure 31: Modeled Streamflow and Patch-level Evapotranspiration vs. Observed Streamflow and Flux Tower ET in P301



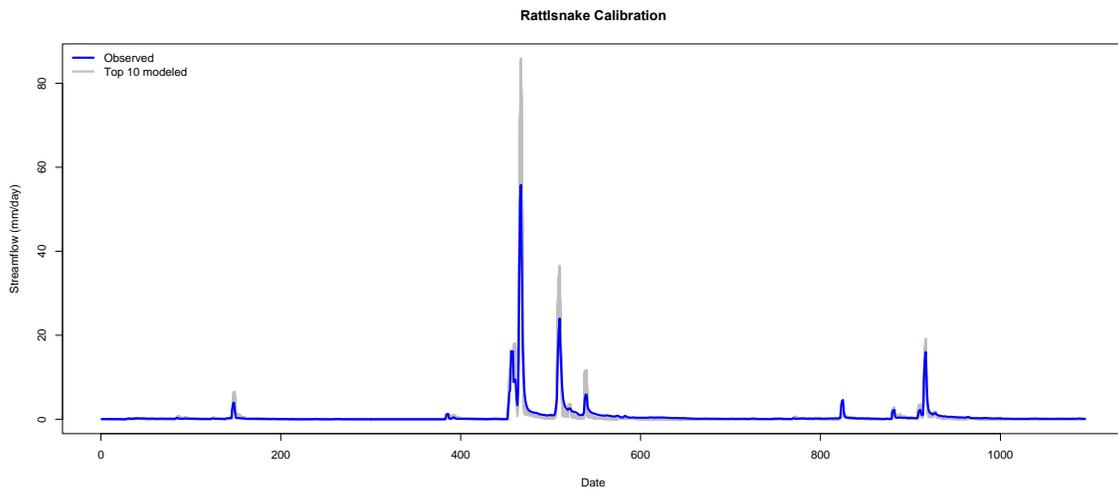
Blue line represents observed flow and gray lines represent modeled output for the ten top parameter sets.

The Rattlesnake watershed had previously been calibrated using non-dynamic chaparral (e.g. vegetation growth was turned off) by Shields and Tague (2014). For this study, values of carbon and nitrogen from the previous calibration were used to initialize calibration using dynamic chaparral, allowing chaparral growth and mortality to respond to inter-annual variations in climatic conditions. The model was ‘spun-up’ for many years to allow the soil carbon and nitrogen stores to stabilize. The model was then calibrated for the wateryears 2004-2006, allowing dynamic vegetation to interact with the hydrology. Over 1350 parameter sets were run, and the final 10 parameter sets chosen were based on streamflow accuracy (Table 10, Figure 32). Calibration against observed streamflow produced good correspondence with NSE, which ranged from 0.71 to 0.76, and bias, which ranged from -2.6% to 6.8% of mean annual streamflow.

Table 10: Calibration Results for Rattlesnake

Objective Function	Performance
NSE daily	0.71 to 0.76
NSE _{log} daily	0.45 to 0.48
% Bias (annual)	-2.6 to 6.8
Accuracy	0.32 to 0.35

Figure 32: Modeled vs. Observed Streamflow in Chaparral-Dominated Rattlesnake Canyon



4.3.3 Model Simulation

Three different change scenarios were examined to identify the effects of vegetation type conversion and climate change on streamflow. Each change scenario was compared to streamflow under baseline conditions. Baseline land-cover conditions in P301 were assumed to be forest while baseline land-cover conditions in Rattlesnake were chaparral shrublands. Baseline temperatures in both watersheds reflected estimates based on historic climate.

In scenario 1, 100% of the watershed area was assumed to be type converted, with no change in temperatures. In P301, vegetation was changed from trees to shrubs, and in Rattlesnake the vegetation was changed from shrubs to grass. This scenario represents the maximum potential change, as it is assumed there would be less impact on streamflow in watersheds with less than 100% type conversion. Scenario 2 assumed no type conversion but a 3°C increase in minimum and maximum daily temperatures. The 3°C temperature increase follows the approach of Tague and Peng (2013) and falls within the range of likely increases in temperature for California by the end of the century. There is currently no agreement amongst current downscaled projections of climate change on precipitation changes in California so no changes in precipitation were assessed in this study. Finally, scenario 3 analyzed the combined impact of

both 100% type conversion and a 3°C temperature increase on streamflow. Soil depth, which represents the storage of infiltrated water for use by vegetation, is highly variable in space and recent work suggests that it may play a critical role in determining evapotranspiration responses to warming (Tague and Peng, 2013). To account for this variable, scenarios were repeated for relatively deep soils (5 m, based on regional estimates (e.g., Bales et al., 2011)) and shallow soils (2.5 m).

For P301, a 55-year (1945-2000) precipitation and temperature record was obtained from a meteorological station near Grant Grove and regressed against the local meteorological data for use during simulation. Tree carbon and nitrogen cycling parameters within the model were based on values for a “standard” conifer taken from RHESSys parameter libraries. Parameters for P301 shrubs differed from conifers through the stomatal conductance-leaf water potential relation (based on field measurements from this study), an increase in leaf turnover for shrubs (also observed in the field campaign), an increase in the proportional allocation of net photosynthate to roots, and a reduction in shrub specific leaf area.

In Rattlesnake, each of the three scenarios was simulated from 1988 to 2009. Carbon and nitrogen cycling parameters for grasses were derived from a grass parameter library in RHESSys.

4.4 Model Output for Chaparral to Grass Conversion in Rattlesnake

The modeled effects of the three simulation scenarios on streamflow, transpiration and evaporation, for both the deep and shallow soil, are summarized in Table 11. The scenarios are compared relative to baseline conditions (i.e. chaparral with historic climate).

Type conversion from chaparral to grass (scenario 1) produced a substantial increase in streamflow, largely as a result of a decrease in transpiration. Lower transpiration from grass reflects its lower LAI, shallower roots and shorter period over which it is actively transpiring. Changes in soil depth from deep to shallow resulted in virtually no change between scenarios, as the shallow rooting system of grasses and intermediate rooting system of chaparral (Hibbert, 1971) did not extend deep enough into the soil to take advantage of the additional water provided by the deeper soil.

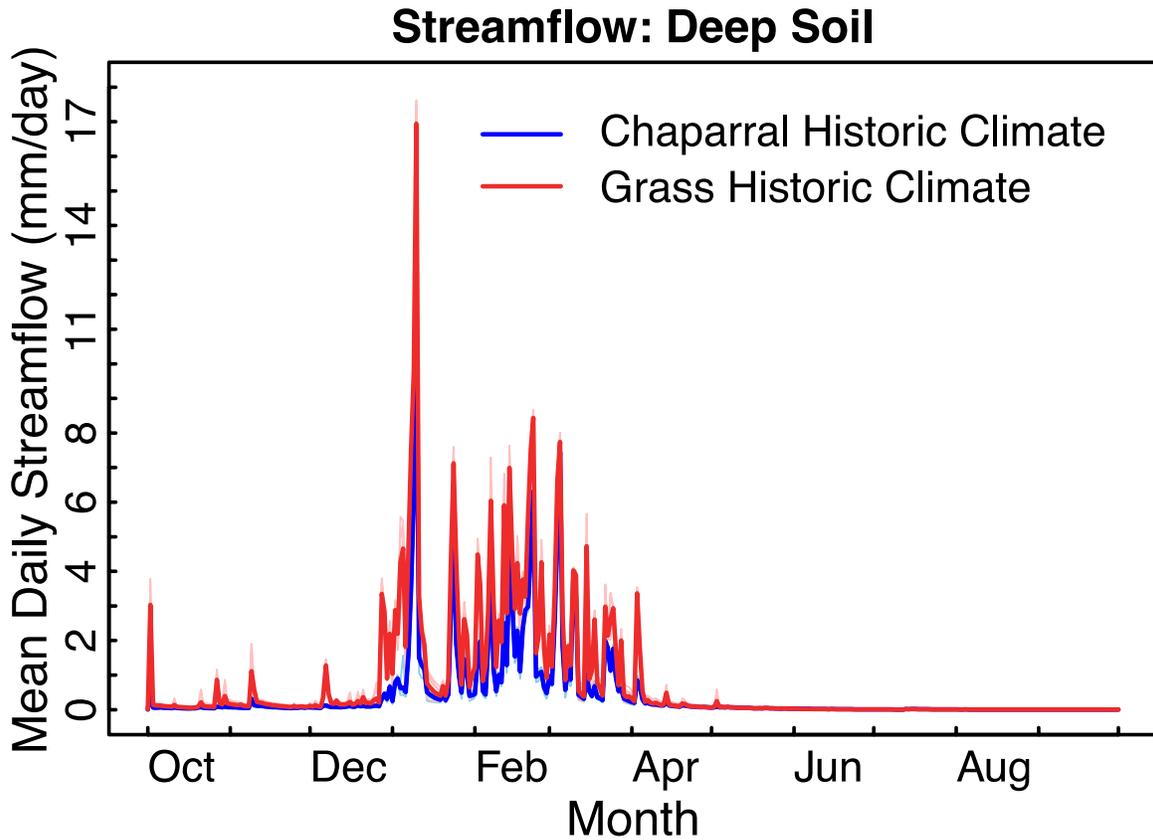
Figure 33 shows the predicted mean daily streamflow averaged by water year day for the baseline and vegetation change scenario under historic climate conditions (scenario 1). Results are shown for each of the 10 parameter sets (faint lines), and the average result across all 10 parameter sets (bold lines). Figure 34 shows the differences in response for chaparral and grass to inter-annual variations in precipitation. Vegetation type change to grass produces greater streamflow in most years and the effects do not vary substantially with precipitation, except in the driest years, where streamflow increases due to conversion are less.

Table 11: Effects of Chaparral to Grass Type Conversion and Climate Change on Watershed Hydrology Separated by Deep and Shallow Soils in Rattlesnake

<i>Deep Soils</i>					
Vegetation Scenario	Climate Scenario	Metric	Mean Annual Streamflow	Mean Annual Transpiration	Mean Annual Evaporation
Chaparral (baseline)	Historic	Total (mm)	163	326	35
Grass	Historic	Total (mm)	298	186	33
		Change (mm)	135	-140	-2
		Change (%)	83%	-43%	-6%
Chaparral	3°C Warming	Total (mm)	168	321	32
		Change (mm)	5	-5	-3
		Change (%)	3%	-2%	-9%
Grass	3°C Warming	Total (mm)	290	193	35
		Change (mm)	127	-133	0
		Change (%)	78%	-41%	0%
<i>Shallow Soils</i>					
Vegetation Scenario	Climate Scenario	Metric	Mean Annual Streamflow	Mean Annual Transpiration	Mean Annual Evaporation
Chaparral (baseline)	Historic	Total (mm)	163	325	35
Grass	Historic	Total (mm)	298	186	33
		Change (mm)	135	-139	-2
		Change (%)	83%	-43%	-6%
Chaparral	3°C Warming	Total (mm)	169	320	32
		Change (mm)	6	-5	-3
		Change (%)	4%	-2%	-9%
Grass	3°C Warming	Total (mm)	290	192	35
		Change (mm)	127	-133	0
		Change (%)	78%	-41%	0%

Change scenarios are relative to the baseline scenario.

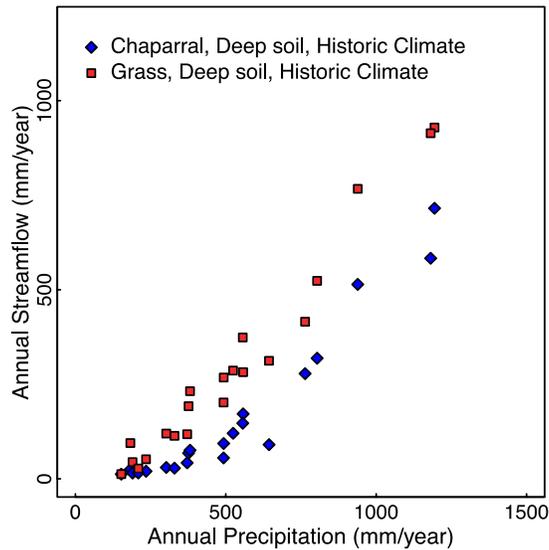
Figure 33: Mean Daily Streamflow Averaged by Wateryear Day for Baseline and Vegetation Change Scenario in Rattlesnake



There was a negligible change to streamflow following a temperature increase of 3°C (scenario 2 and scenario 3) (Table 11). Vegetation in this region is adapted to warm temperatures. Climate warming in this region does not substantially alter atmospheric water demand nor the timing and magnitude of soil moisture recharge. The very small changes in streamflow reflects small declines in transpiration and evaporation by each species. We note however that this project did not account for increases in water use efficiency related to increasing atmospheric CO₂. Previous work has shown that increased productivity associated with higher atmospheric CO₂ in chaparral could ultimately lead to declines in streamflow, and in particular low flows (Tague et al., 2009).

These results indicate that a species type change from chaparral to grass will likely result in an increase in streamflow. For these simulations, a 100% type change was investigated. The magnitude of increase on streamflow may be mitigated if a smaller percentage of the watershed is converted to grass, or depending on the areas within the watershed that undergo the transition.

Figure 34: Comparison of Annual Precipitation and Annual Streamflow in Rattlesnake



4.5 Model Output for Forest to Shrubland Conversion for the Sierra Nevada

4.5.1 Effect of Type Conversion on Streamflow (Scenario 1)

A summary of the effect of tree-to-shrub type conversion on mean annual streamflow, transpiration and evaporation, as well as the mean modeled snow water equivalent (SWE) on April 1, is provided in Table 12. The results are split by deep and shallow soils and show simulation results under baseline conditions (Tree, 0°C change), type conversion from trees to shrubs (shrubs, 0°C change), increase in temperature (trees, 3°C), and both type conversions and an increase in temperature (shrub, 3°C).

For both deep and shallow soils, streamflow increases following type conversion to shrubs. The absolute amount of streamflow change is similar for both soils depths (70 vs. 61 mm change, respectively); however, the percent increase is larger for deeper soils since modeled streamflow under baseline conditions is much smaller for watersheds with deep soils (359mm vs. 537mm). The increase in streamflow following type conversion is produced by a decrease in evaporation. Conversion from trees to shrubs decreases evaporation because shrubs have a lower height, smaller biomass, and smaller LAI, leading to less interception following precipitation events.

Interestingly, overall transpiration increased for the shrubs relative to the trees, despite a reduction in LAI (Table 12). This increase was unexpected, but likely caused by greater water availability in the rooting zone of shrubs during the growing season due to the decrease in losses of intercepted water through canopy evaporation and a slightly later timing of snowmelt. Higher maximum stomatal conductance of shrubs relative to trees, which was observed from field measurements and used to inform the model, also contributed to the high transpiration rates.

Table 12: Effects of Tree-to-Shrub Type Conversion and Climate Change on Watershed Hydrology Separated by Deep and Shallow Soils.

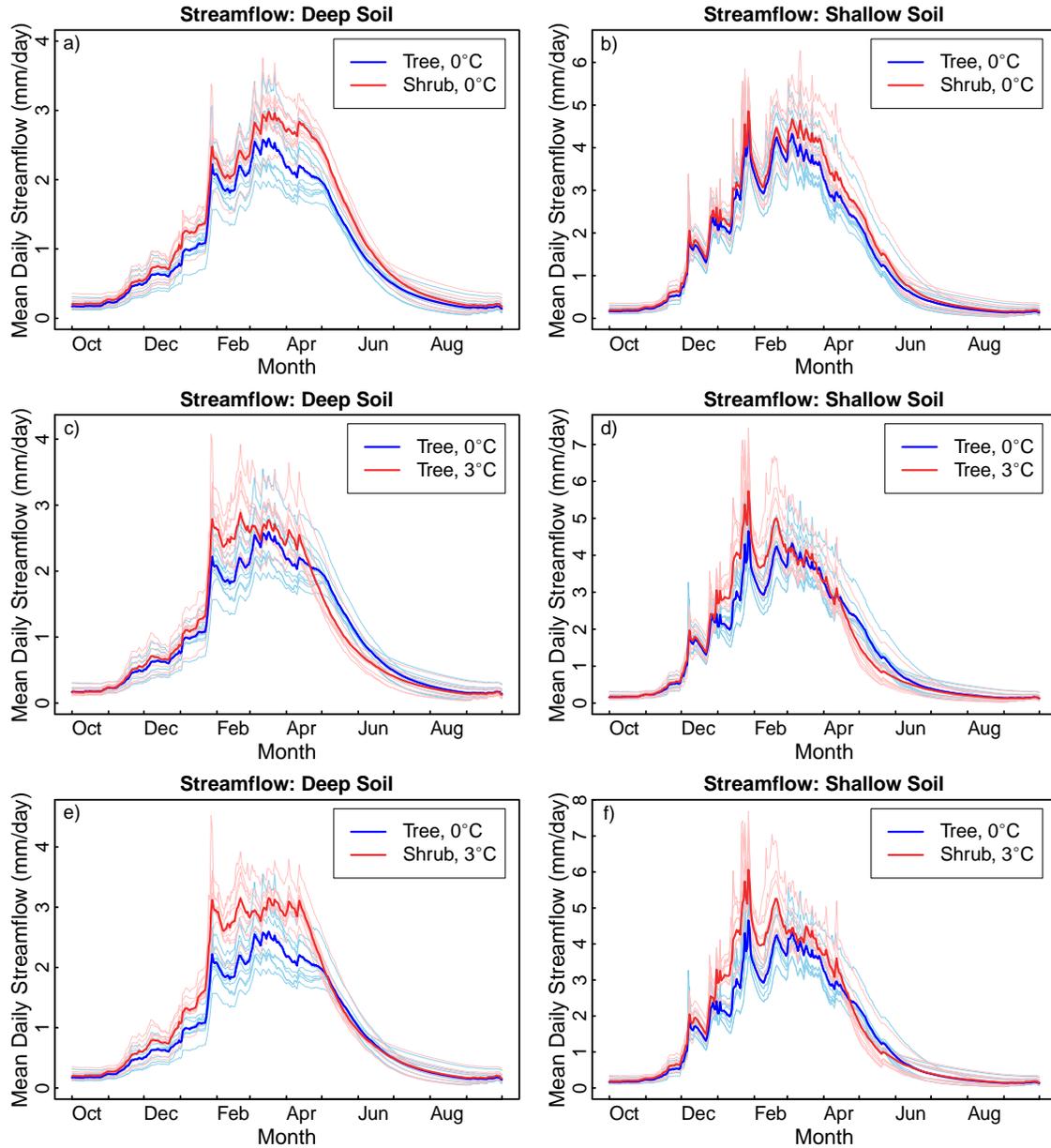
<i>Deep Soils</i>					
Scenario	Metric	Mean Annual Streamflow	Mean Annual Transpiration	Mean Annual Evaporation	SWE (Apr 1)
Tree, 0°C (baseline)	Total (mm)	359	658	281	72
Shrub, 0°C (scen 1)	Total (mm)	429	694	174	83
	Change (mm)	70	36	-107	11
	Change (%)	19%	5%	-38%	15%
Tree, 3°C (scen 2)	Total (mm)	377	679	241	4
	Change (mm)	18	21	-40	-68
	Change (%)	5%	3%	-14%	-94%
Shrub, 3°C (scen 3)	Total (mm)	441	713	142	5
	Change (mm)	82	55	-139	-67
	Change (%)	23%	8%	-49%	-93%
<i>Shallow Soils</i>					
Scenario	Metric	Mean Annual Streamflow	Mean Annual Transpiration	Mean Annual Evaporation	SWE (Apr 1)
Tree, 0°C (baseline)	Total (mm)	537	487	264	72
Shrub, 0°C (scen 1)	Total (mm)	598	518	170	81
	Change (mm)	61	31	-94	9
	Change (%)	11%	6%	-36%	13%
Tree, 3°C (scen 2)	Total (mm)	568	499	220	4
	Change (mm)	31	12	-44	-68
	Change (%)	6%	2%	-17%	-94%
Shrub, 3°C (scen 3)	Total (mm)	621	527	136	5
	Change (mm)	84	40	-128	-67
	Change (%)	16%	8%	-48%	-93%

Change scenarios are relative to the baseline scenario.

Figures 35a and 35b show the simulated mean daily streamflow averaged by wateryear day for the baseline scenario and the vegetation change only scenario, for deep and shallow soils respectively. The thin lines represent the mean daily streamflow for each of the top 10 parameter sets while the thick line represents the ensemble average (where streamflow from each parameter set is combined to produce an average flow). The model results indicate that post-conversion streamflow increases towards the end of the snowmelt season (March, April).

This shift in timing is likely due to an increase in snowpack associated with the shrubs, which extends the duration of snowmelt further into the spring (Figure 36). Modeled SWE increased

Figure 35: Mean Daily Streamflow Averaged by Wateryear Day for Baseline and Change Scenarios for Forest and Shrubs in the Sierra Nevada

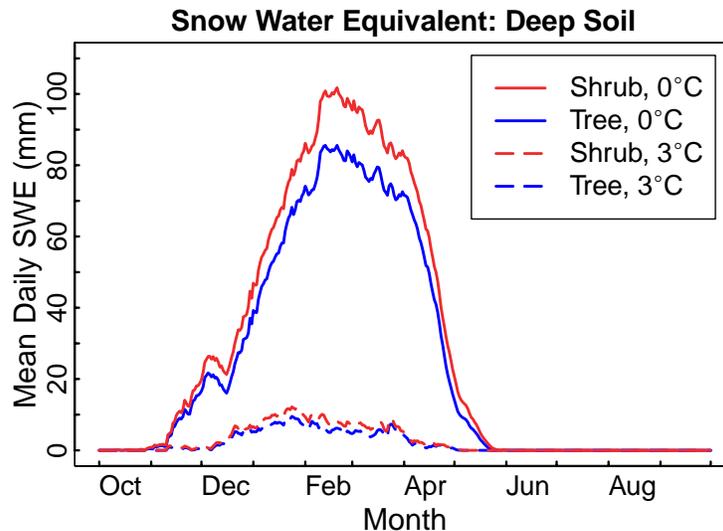


13% to 15% following type conversion (Table 12), a result that conforms with observations in the Sierra Nevada that snowpack increases with decreased vegetation (Harbold et al., 2014).

4.5.2 Effect of temperature change on streamflow (scenario 2)

The effect of a 3°C increase in temperature on Sierra Nevada hydrology in the absence of vegetation change is displayed in Table 12 and Figures 35c and 35d. The most significant change in watershed hydrology is a 94% decrease in April 1 SWE associated with 3°C warming. This dramatic decrease is due to P301 being located in the rain-snow transition zone and baseline winter temperatures spanning the freezing point. The reduction in snowpack decreases the amount of evaporation from the watershed though a decrease in the amount of sublimation.

Figure 36: Snow Water Equivalent (SWE) for Change Scenarios in the Sierra Nevada



Small increases in both transpiration and streamflow were observed with higher temperatures, once again associated with the decrease in canopy evaporation/sublimation leading to additional water inputs to the soil (Table 12). The model also indicates an earlier shift in the seasonal timing of streamflow due to a reduction in snowpack (Figures 35c and 35d, Figure 36). The shift in timing is the product of both an increase in the number of rain vs. snow events during the wet season as well as faster melting of the snowpack when snow does occur (Stewart et al., 2005). Both scenarios contribute to greater runoff in the early wet season and a corresponding reduction in flow at the end of the wet season (Figures 35c and 35d).

Comparing the effects of both vegetation conversion (scenario 1) and temperature changes (scenario 2) on streamflow, the magnitude of streamflow appears to be more sensitive to vegetation change than climate change, while the timing of flow is more sensitive to temperature change. These results suggest that accounting for changes in vegetation communities will be critical for accurately characterizing future hydrologic regimes for the Sierra Nevada.

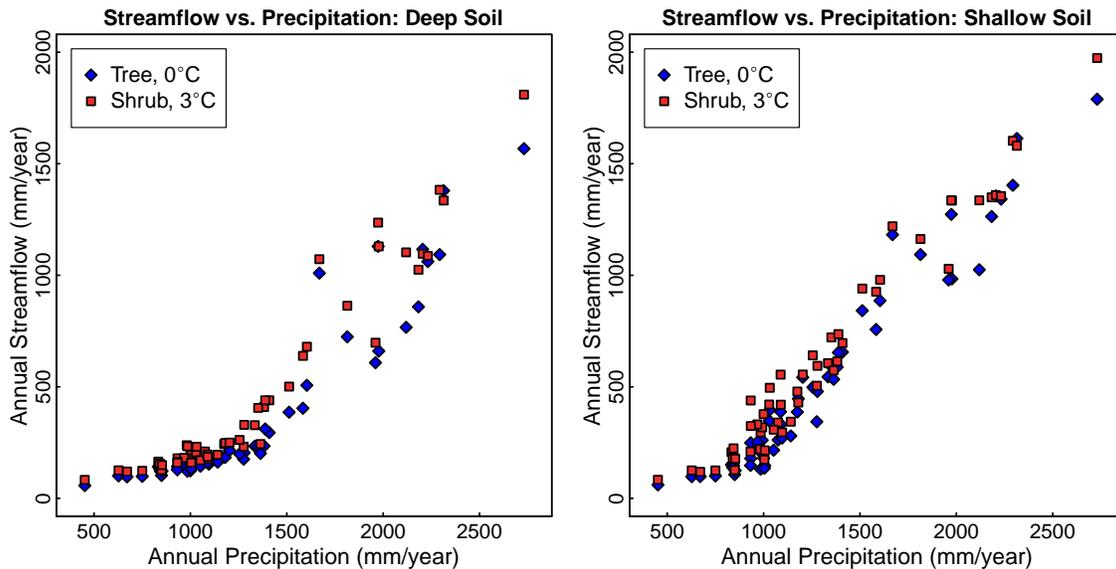
4.5.3 Effect of Type Conversion and Temperature Change on Streamflow (Scenario 3)

The effects of both type conversion and climate change on streamflow are shown in Table 12 and Figures 35e and 35f. For total flow, vegetation change and climate change reinforce one

another, with this scenario producing the greatest increase in streamflow for both deep and shallow soils (23% (82mm) increase for deep soils and 16% (84mm) increase for shallow soils). On the other hand, vegetation change and climate change have opposing effects on the timing of streamflow (Figure 35e and 35f). Conversion to shrubs counteracts the early timing of streamflow caused by warmer temperatures. For deep soils, the increase in streamflow production combined with the later timing of streamflow with vegetation change causes the seasonal recession curve to coincide with the baseline curve. For shallow soils, the earlier shift in timing due to increased temperatures dominates, potentially increasing water stress later in the dry season.

Modeled annual streamflow was plotted against annual precipitation to show the year-to-year variability in streamflow response to shrub conversion and increases in temperature (Figure 37). These plots reveal that post-conversion streamflow change is greatest during wet years, with limited changes during drier years. The precipitation threshold at which streamflow response is more strongly affected by vegetation and climate change varies by soil depth, with a threshold of approximately 1400mm for deep soils and 1000mm for shallow soils. These results suggest that dry watersheds with shallower soils may be more sensitive to vegetation change than dry watersheds with deep soils.

Figure 37: Comparison of Annual Precipitation and Simulated Annual Streamflow in the Sierra Nevada



The results shown in this section indicate that streamflow is likely to increase following type conversion from trees to shrubs in regions with climates similar to the rain-snow transition climate of the Southern Sierra Critical Zone Observatory. However, more investigations will be needed to determine if the results are applicable to all rain-snow transition areas, or even the Sierras as a whole. Vegetation conversion across various mountain ecosystems is likely to be highly variable both spatially and temporally. Further, the modeling scenarios in this study can

be viewed as end-member scenarios, as 100% of vegetation cover was altered in the watershed. Streamflow response to tree-to-shrub type conversion may not scale linearly when smaller percentages of a watershed are subject to type conversion. The magnitude of the streamflow change results may also vary under alternative model parameterizations, particularly with varying levels of LAI for the trees and shrubs. Finally, the elevations within the Sierra Nevada that produce the largest percentage of runoff are located above than the rain-snow transition zone (Goulden and Bales, 2014). Understanding the potential for type conversion at these higher elevations will be necessary to gain a comprehensive understanding of how type conversion may affect Sierra Nevada waters supplies.

4.6 Implications of Forest to Shrubland Conversion for Reservoir Capacity

The effect of tree-to-shrub type conversion on reservoir capacity was investigated by 1) modeling post-conversion streamflow change in a medium sized watershed, Big Creek, and 2) comparing post-conversion streamflow changes on the functioning of a hypothetical reservoir located at the outlet of the watershed. Big Creek is a south-flowing river whose watershed encompasses P301 as a tributary on its east flank (Figure 28). The watershed is 65.7 km², and elevations extend from 957 m to 2344 m. Vegetation in the watershed ranges from a ponderosa pine forest with oaks at the lower elevations to a mixed conifer forest at upper elevations.

Big Creek is an ungauged stream, and no streamflow data are available to directly calibrate the RHESSys model. Instead, parameter sets from the top calibrations in P301 were transferred to Big Creek, similar to a procedure used by Tague et al. (2013). Despite the inability to directly calibrate RHESSys in Big Creek, it was expected that the relative differences between change scenarios to still be representative of the results had the model been calibrated to Big Creek. Streamflow in Big Creek was simulated in the same manner as simulations in P301.

Meteorological data were obtained from the Grant Grove station for the period 1945 to 2000. Three change scenarios were once again used to investigate the effect of 1) tree-to-shrub type conversion, 2) a 3°C temperature increase and 3) both type conversion and a temperature increase combined. For simplification, only one soil depth of 3m was used for the watershed.

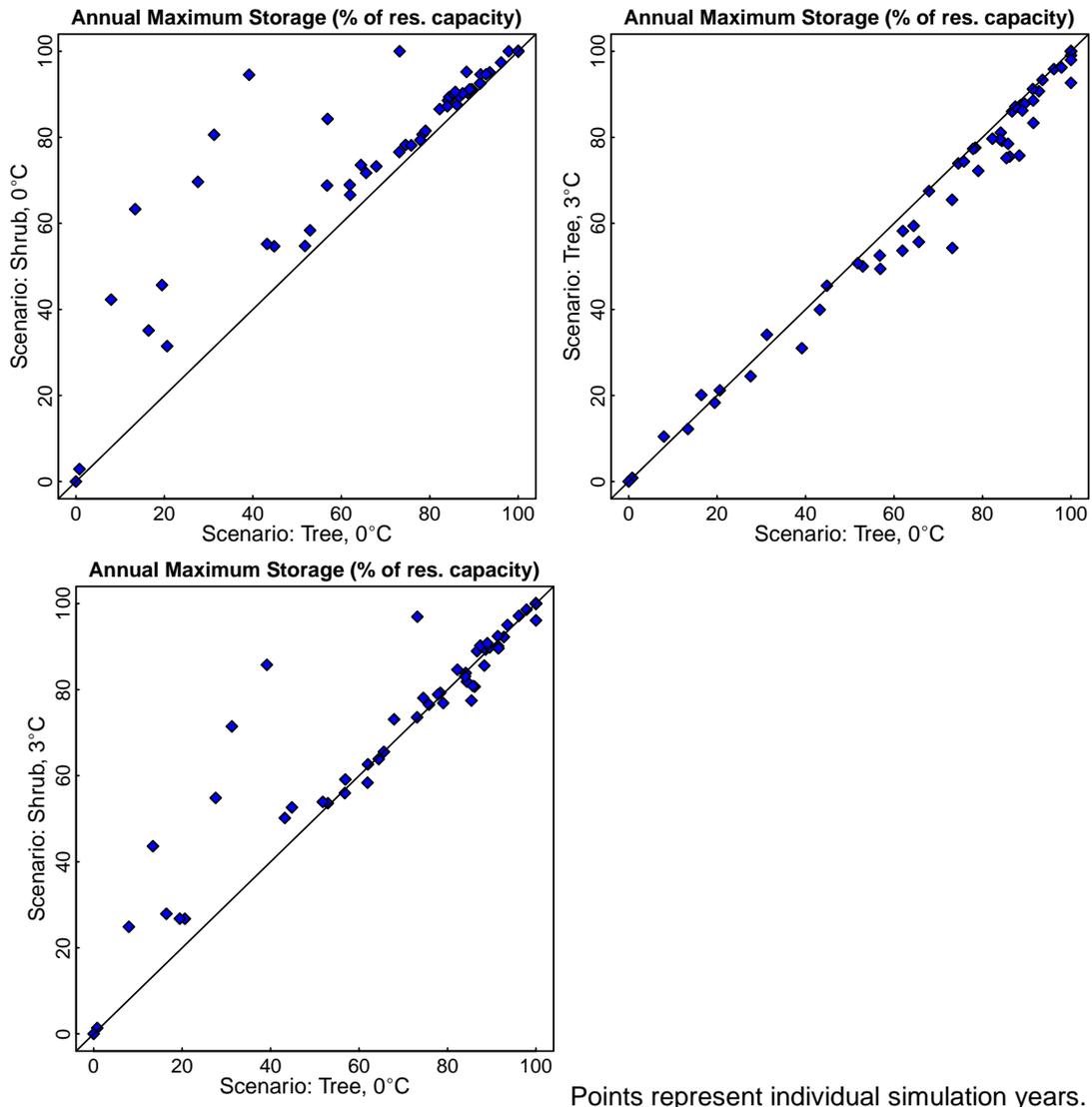
A parsimonious reservoir model was used to test the effects of the three change scenarios on reservoir capacity. The reservoir model operates at a daily time step using RHESSys simulated daily flow for the Big Creek watershed as inflow. Key characteristics of the reservoir model include maximum storage, minimum storage level, minimum flow requirement and the operating rule curve. In the case of conflict between minimum storage levels and minimum outflows, priority was given to maintaining minimum flows.

The parameter values for the reservoir were selected manually to provide plausible scenarios for comparing the impacts of different vegetation types on reservoir storage levels. For the runs in this section, the maximum storage capacity of the reservoir was assumed to be 30% of the modeled mean annual streamflow out of Big Creek. A monthly rule curve was developed for the reservoir based on similar rule curves in Sierra Nevada reservoirs. Maximum allowable storage for a given month was 50% of total reservoir capacity for January-March, 70% for April,

90% for May, 100% for June through October, 75% for November, and 50% for December. Minimum storage was set at 10% of reservoir capacity and minimum outflow (m³/s) was set at 50% of mean annual flow.

Figure 38 shows the effect of the three change scenarios on the annual maximum storage in the reservoir, i.e., the highest storage level obtained in the reservoir for a given year. Following type conversion from trees to shrubs (scenario 1), annual maximum storage increases in the reservoir. However, the increase is generally minimal during wet years because there is little excess winter storage capacity for the extra streamflow generated following type conversion. In dry years when winter storage levels are below the capacity defined by the rule curve, storage is available to capture the additional inflow, and this is reflected in an increase in annual maximum storage.

Figure 38: Comparison of Annual Maximum Reservoir Storage for Baseline (Tree, 0°C) and Three Change Scenarios



Climate change in the absence of vegetation change (scenario 2) produces a reduction in annual maximum storage because more water moves through the reservoir during the winter when storage is limited (Figure 38). For example, if snowpack that would normally persist until April melts in March under a 3°C warming scenario, this water may not be captured by the reservoir since the maximum allowed storage capacity is smaller in March than in April. For scenario 3, the increases in annual maximum storage that were produced following conversion to shrubland were reduced compared to scenario 1, as the 3°C increase in temperature caused an earlier shift in streamflow timing and partially canceled out the effect of increased storage with shrubland.

In addition to changes in reservoir storage, the study examined the impact of vegetation and climate change on in-stream requirements. The frequency at which outflow from the reservoir dropped below the minimum flow requirement was assessed for the month of August. For the baseline scenario on Big Creek, the reservoir was able to meet the monthly flow requirement during the month of August for 46 out of 55 years (84%). For scenario 3, the reservoir was able to meet monthly flow requirements for 50 out of 55 years (91%), with a substantial increase in outflow relative to the baseline scenario in an additional 2 years.

Despite moderate increases in mean annual streamflow (16-23% increase in P301) following type conversion from forest to shrubs, the effect of type conversion on annual maximum storage and in-stream requirements was limited to dry years. Additional post-conversion streamflow during wet years passes through the reservoir and cannot be held for future use. Consequently, increases in hydropower generation will likely be limited to the winter months when demand for hydropower is often lowest.

Finally, it should be reiterated that for climate change, only a temperature-increase scenario was analyzed in this study, as the effect of climate change on precipitation in California remains uncertain. An increase or decrease in precipitation has the potential to modify the study results and may need to be investigated in future studies.

CHAPTER 5: Conclusion

California faces an uncertain future in terms of water, energy and climate. The motivation for this project is a basic concern over how climate change, fire, and vegetation will interact to influence watershed hydrology and stream flow into the future. An understanding of possible scenarios is needed for adaptation planning and improved ecosystem and water supply management. Models are one tool for simulating possible futures; however, such models must be grounded in actual measurements. This research project has thus assessed fire-driven vegetation-change, water use measurements on potential dominant species, and scaling of water use from the leaf- to landscape-scale. It has incorporated a sophisticated ecohydrological model to evaluate how different climate and vegetation scenarios influence water yield and ultimately reservoir levels.

Short interval fire has been implicated in replacement of Southern California chaparral with invasive grasses (Keely and Brennan, 2012; Lippitt et al., 2013) with unknown consequences for landscape hydrology. The research reported in Chapter 2 evaluated both the roles of altered fire regime in causing vegetation change and the ecophysiology of different dominant species that might be influenced by fire and climate change in chaparral. Analyses of vegetation change were performed at two spatial scales, fine spatial resolution aerial imagery for Los Angeles and Ventura Counties, and coarser resolution Landsat TM data for coastal southern California stretching from Santa Barbara County to San Diego County. Both analyses indicated that vegetation change from shrub to grass cover is not common following short interval fires. Researchers compared “once burn” and “twice burn” polygons and found no significant differences in percent shrubland vegetation cover following one fire versus two fires within a five year period. Instead, aspect was a strong predictor of vegetation change: percent grass cover was observed to increase following fire on south facing aspects. In the remote sensing imagery analysis, also did not find extensive type conversion of shrubland to grassland, suggesting that type conversion of shrubland to grassland may be a more localized or spatially patchy phenomenon. Simple linear regression indicated that only elevation was strongly correlated to observed variation in vegetation recovery (and conversion of shrubland to grassland) across the once versus twice burned pairs. At lower elevation, non-sprouting species are more common and thus the vegetation is likely more vulnerable to type conversion. The two low elevation sites demonstrating the most change were both in the Santa Monica Mountains, which may be more vulnerable to changes in vegetation cover due to intensive human access, greater occurrence of less fire-tolerant coastal sage scrub species, and the predominance of obligate seeding species. While broad changes in vegetation type were not found in the remote sensing and the aerial imagery analysis, species composition was not assessable. Further study is needed to determine relationships between fire-return interval, shifts in species composition, and loss of ecologically important chaparral species.

In the final section of Chapter 2, researchers examined the responses of three common chaparral shrub species to water availability during an extreme drought to better understand how these

shrubs may impact watershed scale hydrologic function, how shifts in dominance might influence watershed yield and whether mortality of important species would occur during extreme drought. Two of the three species reached water potentials values (Ψ_{pd}) at or below their threshold of water potential beyond which hydraulic failure is likely to occur (Ψ_{P50} values). Yet they did not die. The third species, an important dominant of xeric chaparral, was able to maintain higher water potential and higher stomatal conductance throughout the drought than the other species, suggesting remarkable drought adaptation. Water availability increased sharply in all three shrub species immediately after both small rain and large fog events. These results suggest that dominant fire-adapted chaparral species will slow or stop photosynthesis and close their stomata as water availability becomes highly limiting and are more responsive to moisture availability than previously assumed. Despite some differences between the dominant species, researchers concluded that during extreme drought, the ecophysiological response of three common chaparral species was relatively similar as water availability declined, which indicates that species-level effects on watershed hydrological function may be less important than changes in plant functional types, i.e. shrub to grassland conversion. Comparisons of watershed yield under different shrubs species, and shrubs versus grasses are discussed later.

The first section in Chapter 3 examined post-fire vegetation recovery in mixed conifer and red fir forests in the Sierra Nevada using a vegetation index time series calculated from Landsat TM data. Given the expected increases in fire severity and drought trend as the climate changes in western United States ecosystems, we focused on the effects of fire severity and post-fire climate on forest change. High and moderate severity fire demonstrated more prolonged impacts on vegetation cover (as measured by the index) compared to low severity fire, with red fir forest showing more persistent reduction in vegetation cover relative to mixed conifer forest. Environmental factors were also found to be important predictors of vegetation recovery six years following fire. The modeling results indicated that mixed-conifer forests were sensitive to post-fire drought, while red fir forests were sensitive to post-fire temperature anomalies. The sensitivity of mixed conifer forest to wet season precipitation anomaly indicates that short-term (the current California drought) and long-term (climate change) climatic trends may have a detrimental impact on post-fire tree recovery in mixed conifer forest. The study suggests that adaptive management guidelines for different forest ecosystems might be necessary under several climate change scenarios, since mixed-conifer and red fir forests demonstrated varying responses to post-fire climate variables. Post-fire management of forested watersheds should thus take into account forest type and climatic conditions following fire. For example, post-fire recovery may be reduced when drought follows fire, requiring more active reforestation if maintaining forest cover in specific locations is a management goal. These results, combined with longer term monitoring, may be useful to incorporate into a local forest gap model or for calibrating ecosystem simulation models to gain a better understanding of the interactions between climate, disturbance and vegetation dynamics.

The final section in Chapter 3 investigated how hydrologic function of the watershed-scale may be impacted by projected shifts in vegetation dominance (trees to shrubs) in a future climate. The researchers focused on studying the ecophysiological function of dominant tree versus

shrub species at the rain-snow transition zone in the southern Sierra Nevada. This study was conducted during the driest year recorded over the past 100 years in California; therefore, the observations should provide especially useful insight into how vegetation types may (or may not) tolerate drought conditions in the future. The authors cannot conclude from their results that shrubs use more water than trees. However, since the total leaf area is greater for trees than shrubs, it appears that fluxes of water from the soil to the atmosphere would be greater in a shrub-dominated compared to a tree-dominated landscape. Such a result has important implications for the partitioning of sub-surface water to transpiration versus streamflow in the future. The outcome of this field study refined estimates of essential parameters in an ecohydrologic model that could address how vegetation type conversion would impact streamflow.

In Chapter 4, streamflow was analyzed in both a southern California chaparral watershed and in a Sierra Nevada mixed conifer forest watershed for three change scenarios: 1) vegetation type conversion only, 2) a 3°C temperature increase and 3) both type conversion and a temperature increase combined. In southern California, conversion from a chaparral-dominated watershed to a grass-dominated watershed (scenario 1) resulted in a substantial 78% increase in mean annual streamflow following 100% type conversion. This increase was greatest during wet years; with minimal change observed during dry years. Streamflow in southern California was insensitive to higher temperatures (scenarios 2 and 3). Streamflow was also insensitive to which chaparral species were dominant which is consistent with the similarity in ecophysiological function reported in chapter 2.

In the Sierra Nevada, 100% conversion from forest to shrubland (scenario 1) increased mean annual streamflow between 11% and 19% compared to historical conditions, with watersheds having deeper soils being most sensitive to type conversion. This increase in streamflow was the result of a reduction in evaporation (e.g., interception) associated with smaller shrub biomass compared to forests. Post-conversion changes in streamflow were greatest during wet years, with limited changes during drier years. Under scenario 2 (3°C temperature increase), a dramatic reduction (~93%) in snowpack (at the current rain-snow transition zone) contributed to a slight increase in mean annual streamflow (5%). This effect on streamflow was much smaller than that of type conversion, and illustrates the importance of accounting for vegetation change when predicting future impacts of climate change on streamflow. Under the type conversion-temperature increase scenario (scenario 3), mean annual streamflow increased between 16% and 23% over historical conditions.

Post-conversion streamflow changes in the Sierra Nevada were linked to a reservoir model to investigate potential impacts on hydroelectric capacity. The results showed that annual maximum reservoir storage increased following forest-to-shrub type conversion (scenario 1), however the increase was generally minimal during wet years due to insufficient reservoir storage capacity during the winter. During dry years, storage was available to capture the additional inflow. An increase in temperatures in the absence of vegetation change (scenario 2) had the opposite effect on annual maximum storage, with a slight decrease in annual maximum storage due to an earlier shift in winter inflow. For the type conversion with a 3°C temperature increase (scenario 3), the impact of type conversion on annual maximum storage was dominant,

with the increase in annual maximum storage slightly reduced compared to scenario 1. These modeling results highlight that unless there is sufficient reservoir capacity to capture additional winter flows generated by type conversion, increases in hydropower generation will likely be limited to the winter months when demand for hydropower is often lowest.

Wildfire has been implicated in converting vegetation from one life form to another, such as from forest to shrubland or shrubland to grassland. Human ignitions and drought are likely to continue to promote wildfire occurrence in California. As vegetation changes occur or recovery is slowed by post-fire drought, as evidenced in the Sierra Nevada research reported here, soil and watershed hydrology will be affected. This is because plant life forms differ in the way they use water leading to changes in runoff, streamflow, reservoir storage, and ultimately to end-uses such as hydropower. Climate change is expected to dramatically increase the frequency (chaparral) and intensity (Sierra Nevada) of wildfire, as well as to induce greater water stress on plants both pre and post-fire. Anticipating the potential outcomes of the interactions of this complex set of processes is extremely challenging. This study is among the first to analyze this complex interaction holistically across two different dominant ecosystem types in California. The study found limited evidence of vegetation type conversion from single short-interval fire occurrences in southern California but it did identify a general increase in grasses with fire particularly on south facing slopes. If conversion from shrubland to grass eventually occurs, this is likely to result in much greater runoff into streams. Sediment yields with conversion were not assessed but are a concern. In contrast, the study found reduced recovery of mixed conifer forests in the Sierra Nevada related to high fire intensity, post fire drought and local scale climate. Reduced recovery of trees means more cover of shrubs and such conversion leads to modest increases in runoff, which is further boosted by temperature increases at a level projected for climate change. The degree of change in runoff varies between wet and dry years. Greater understanding of ecosystem interactions could lead to an approach for improved forecasting for reservoir management and hydroelectricity generation, helping Californians adapt both to existing conditions and to the effects of climate change on the electricity system.

GLOSSARY

Term	Definition
Accuracy	An objective function combining the individual objective functions NSE, NSE _{log} and Percent Error.
Annual maximum storage	The highest storage level obtained in the reservoir for a given year.
Aspect	The compass direction that a slope faces. North: 0° and 360°; East: 90°; South: 180°; West: 270°.
Georectification	The digital alignment of a satellite or aerial image with a map of the same area. A number of corresponding control points are marked on both the image and the map, which become reference points in the subsequent processing of the image.
LAI	Leaf Area Index is a measure of the leaf area in a plant canopy.
Mesic	A type of habitat with a moderate or well-balanced supply of moisture, in comparison to a hydric and xeric habitat.
Nash-Sutcliffe Efficiency	An objective function emphasizing the fit of peak flows.
Nash-Sutcliffe Efficiency (logged)	An objective function emphasizing the fit of moderate flows.
NBR	Normalized Burn Ratio was defined to highlight areas that have burned and to index the severity of a burn using remote sensing data
NDVI	Normalized Differenced Vegetation Index was defined to quantify the amount of green vegetation of the observed areas using remote sensing data.
Objective Function	A measure used to evaluate the goodness of fit between modeled streamflow and observed streamflow.
Panchromatic	Black and white imagery exposed by all visible light.
Predawn Leaf Water Potential	Plant water availability at the rooting zone.
Shapefile	A geospatial vector data format for geographic information system software containing point, line, or polygon features.

Solar zenith angle	The angle measured from directly overhead to the geometric center of the sun's disc, as described using the observer's local horizon as the fundamental plane.
Stomatal Conductance	The degree to which a stomata are open or closed.

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