UNIVERSITY OF CALIFORNIA

Santa Barbara

Effects of Short Interval Wildfires on Southern California's Wildland Communities
Using Historical Aerial Photographs

A thesis submitted in partial satisfaction of the

requirements for the degree of Masters of Arts

in Ecology, Evolution, and Marine Biology

by

Stephanie Akemi Ma

Committee in charge:

Professor Carla M. D'Antonio, Chair

Professor Philip E. Dennison, University of Utah

Dr. Max A. Moritz, University of California, Berkeley

Professor Ryoko Oono

June 2016

The thesis of Stephanie Akemi Ma is approved.

Philip E. Dennison

Max A. Moritz

Ryoko Oono

Carla M. D’Antonio, Committee Chair

June 2016

Effects of Short Interval Wildfires on Southern California's Wildland Communities
Using Historical Aerial Photographs



by

Stephanie Akemi Ma

AKNOWLEDGMENTS

First and foremost, thank you to Dr. Carla M. D'Antonio for her continuous supply of support, encouragement, and honest feedback. She offered me the opportunity to earn my M.A. and I am still only beginning to understand my good fortune having her as my mentor and advisor.

Thank you to the D'Antonio and Dudley Labs for their support and friendship throughout the years. I would especially like to thank Viviane Vincent for welcoming me into the lab and providing me with a community when I was greatly in need of one.

Thank you to Dr. Philip E. Dennison and Dr. Max A. Moritz for providing guidance and context for my research. I began this project with barely any geography experience or knowledge of wildfires and they helped me expand my proficiency in both. Thank you to Dr. Ryoko Oono for providing support and feedback.

Thank you also to those that helped get this project (and me) off the ground: Dr. Seth H. Peterson for numerous training sessions in ENVI; Nathan Emery for editing the fire data and for developing initial methods; the VIPER Lab and Dylan Parenti for providing the network support to perform this research; Jon Jablonski, Ann Hefferman, and Deborah Lupo at UCSB's Mapping and Imagery Library for providing countless historical aerial images; Jeremy Dean, Emma Burley, Soumya Suresh, Chase Allred, and Travis Madden for their volunteer hours because, without their help, I would still be selecting ground control points; and Matt Plummer for his expertise in ArcGIS and his youthful attitude in the lab.

I am incredibly grateful for my funding sources acquired by Dr. Carla M. D'Antonio: the California Energy Commission (500-10-045) and the Schuyler Endowment Fund. With their support I was able to focus on my research and dedicate myself to the science.

Thank you to Samuel E. Prentice III for scientific discussions over food, drinks, and, sometimes, Skype. Thank you for always reminding me of the "big picture" and for helping me discover the story within the science.

Thank you to Stephanie R. Arguelles for her friendship, support, and laughter to get through all the ups and downs of life.

To my parents, Dr. Leonard and Sanae Ma, who have stood behind me every step of the way with love and patience – thank you. Without them, the journey of both school and life would be much more challenging.

Finally, thank you to Dr. Erik A. Lucero for his unending love and support. I am a better person and a better scientist because of his influence. My life is so much more balanced and adventurous shared with him and his infectious enthusiasm encourages me to pursue my goals and all things that make me smile.

*This thesis is dedicated to my grandfather and grandmother,*

*Suenobu and Shizuye Yamasaki*

Ji-chan was my model for working hard and pushing forward to become all that I am capable of in life. Ba-chan has shown me how to live with unending strength and beauty. Discussions at the dinner table and sharing old photo albums will always be among my favorite memories.

ABSTRACT

Effects of Short Interval Wildfires on Southern California's Wildland Communities Using Historical Aerial Photographs

by

Stephanie Akemi Ma

Wildfire return intervals are expected to decrease in shrubland communities across southern California due to increasing anthropogenic fire ignitions and climate change. These shortened fire intervals may initiate a positive feedback, placing native chaparral species at risk of replacement by alien annual grasses. This shift is predicted because many chaparral species require multiple years to recover from a wildfire disturbance (i.e., to replenish the seedbank or to replenish underground carbohydrate reserves) and if a second wildfire occurs in quick succession, these species may experience a severe population decline or even extirpation as new seedlings or young resprouts are killed.

After generating a wildfire occurrence map, I selected twelve polygons that experienced two wildfires within five years to evaluate vegetation change following a short-interval fire. All polygons were located in Ventura or Los Angeles County, California and spanned a temporal range from 1956 to 2003. These polygons were then compared to adjacent polygons that experienced only one wildfire within the same five-year period. In order to capture prefire vegetation conditions, historical aerial photographs (HAPs) were selected as-close-to before the first wildfire as possible. In order to capture the maximum postfire growth, HAPs were chosen to be no less than six years following the second wildfire. Prefire and postfire images were georectified and used to calculate ground cover using five community types: chaparral, alien annual grass, sage scrub, tree or bare ground/exposed rock.

To determine the role of regional moisture gradients and environmental conditions in predicting vegetation change I investigated aspect, location in relation to the Santa Ana wind corridors, distance from the coast, time since fire, and prefire cover values as potential predictors of the strength and direction of vegetation responses. Results showed no significant differences in vegetation cover from short-interval wildfires compared to adjacent single wildfires. Prefire vegetation cover was highly correlated with postfire cover. Chaparral and sage scrub cover showed strong trends in relation to the time since fire, especially following a single wildfire: positive vegetation change for chaparral and negative vegetation change for sage scrub. Location in relation to the Santa Ana wind corridors and distance from the coast were not found to be significant factors of vegetation change. By contrast, 'aspect' correlated with significant differences in vegetation cover, regardless of wildfire history, for chaparral and sage scrub communities: chaparral cover declined and sage scrub cover increased after wildfire on north, but not south, aspects. This study did not find evidence of chaparral loss strictly as a result of a single short-interval wildfire. I propose that additional factors, such as aspect and initial community cover (extent), may be equally or more important than a single short-interval fire when predicting vegetation changes following wildfire in southern California.

Keywords: chaparral, alien annual grass, short interval wildfire, historical aerial photographs, southern California

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

# I. INTRODUCTION

Fire-prone ecosystems in the United States are experiencing alterations from their historical fire regime due to increased human influence (Syphard et al. 2007a, Nowacki and Abrams 2008). Fire frequency, or the interval time between fire events, is the most direct way humans can alter a fire regime. For example, fire suppression has lengthened the interval time between fires in the northern Rockies (Barrett and Arno 1982), in western Washington (Everett et al. 2000), and in the Eastern US (Nowacki and Abrams 2008). As a result, fire-adapted tree species have been replaced by fire-sensitive, shade-tolerant ones. A shorter fire interval can also place fire-prone ecosystems at risk. In southern California, a fire return interval shorter than what is considered to be historical, can result in native shrub species being replaced by alien annual grasses (Haidinger and Keeley 1993, Zedler et al. 1983, Keeley 2001).

In these southern California chaparral ecosystems, the historical fire return interval is approximately 20-60 years (Keeley 1987, Keeley et al. 2004) and in some locations may be as long as 150 years (Syphard et al. 2006). Fires are typically crown fires, burning all of the above-ground biomass, in this classic Mediterranean climate region (Hanes 1971). Chaparral shrubs then return to prefire canopy cover generally within the first decade (Hope et al. 2007, Peterson and Stow 2010) and to prefire stand conditions within the second decade following fire (Hanes 1971).

In contrast, the current fire return interval can be much shorter due to increased anthropogenic fire ignitions caused by increasing human populations (Keeley and Fotheringham 2001) and an expansion of the wildland-urban-interface (Syphard et al. 2007b). Climate change is also expected to shorten mean fire return intervals as southern

California becomes warmer (Krawchuk and Moritz 2012). Furthermore, with the introduction of alien annual grasses, chaparral communities can be driven toward a new successional trajectory (D'Antonio and Vitousek 1992) leading to the extirpation of many shrub species and the expansion of opportunistic alien annual grasses (Brooks et al. 2004, Keeley and Brennan 2012).

Unintentional shifts from chaparral dominated communities to alien annual grass dominated communities due to a short interval fire sequence was proposed in the 1980's by Zedler et al. (1983) when two wildfires burned portions of San Diego County in 1979 and 1980, the latter fire reburning a portion of the former. Zedler et al. (1983) found a reduction in density of key chaparral shrub species including *Ceanothus oliganthus* and *Adenostoma fasciculatum* in the initial postfire year in the reburned area. They also noted that *Bromus* and *Avena* were highly abundant without being seeded into the site. Haidinger and Keeley (1993), also in San Diego County, monitored *A. fasciculatum* regrowth following a short interval wildfire (1986, 1991) and likewise found a reduction in resprout and seedling density compared to sites that burned only once (1991). They also found an increase in weedy species such as *Schismis barbatus* and *Brassica nigra*. Later on, Keeley and Brennan (2012) monitored regrowth following a short fire interval (2003, 2007) and again recorded a reduction in *A. fasciculatum* and *Ceanothus tomentosus* density and an increase in *Bromus madritensis*, an invasive alien annual grass, at sites with the four-year fire interval compared to sites that had a nine- to thirty-one-year interval.

A decline in chaparral cover and an increase in annual grasses have also been predicted with vegetation models. Simulating three fire-regime treatments (60, 30, 15 years), Syphard et al. (2006) found the shortest fire-regime treatment (15 years) led to the highest amounts of annual grass cover and lower amounts of chaparral cover compared to the longest treatment (60 years). They further predicted direct conversion from chaparral to annual grasses under the shortest fire regime treatment (15 years).

Some studies, however, indicate a natural tendency for chaparral shrubs to regain dominance over annual species, or at least more complex interactions of vegetation communities under differing conditions, and several practitioners have noted how difficult it is to eliminate chaparral (Bentley 1967, Rosario and Lathrop 1974, Fuhrmann and Crews 2001). Keeley et al. (2006) found most chaparral species recovered to prefire conditions five years after wildfire in San Diego County. From the same wildfire, Keeley et al. (2005a) also observed a continued increase in shrub and subshrub cover and a decline in annual species cover over time, suggesting a correlation between total shrub cover and time since fire. Furthermore, Syphard et al. (2007b) predicted conversion from alien annual grass cover back to chaparral cover under a fire frequency of 60 years suggesting that with enough fire-free years, chaparral could reestablish after annual grass invasion.

Thus, although loss of individual shrubs and declines in shrub seedling recruitment have been observed on very short postfire time scales, the question remains as to what happens to chaparral on a longer time scale following a short interval fire? Jacobson et al. (2004) compared long (≥12 years) and short interval fires (≤6 years), in Los Angeles County, with seven to twenty-five fire-free years postfire. They measured present-day functional group densities in the field and found significantly fewer chaparral species that rely on seeds for recruitment and a significant increase in coastal sage scrub species (drought deciduous shrubs and subshrubs) at sites that experienced a short interval fire. Meng et al. (2014) used 30-meter multispectral Landsat TM data to calculate the difference in chaparral cover across southern California between paired sites of long and short (<8 years) interval wildfires with one to eight fire-free years postfire. Their results found no strong overall trends in shrub reduction due to a short interval fire (i.e., half of the sites increased in chaparral cover following a short interval fire while the other half either remained the same or chaparral cover declined), but they did find a strong correlation between elevation and postfire chaparral regrowth. Specifically, chaparral communities at lower elevations were more susceptible to reduced recovery, which coincides with elevations where sage scrub and alien annual grasses were abundant, suggesting that conversion could be occurring but only in some portions of the landscape.

While plot level data (e.g., Jacobson et al. 2004) offer insight into change at very local scales, Landsat data (e.g., Meng et al. 2014) are taken at a very large scale and are only available since 1983, obscuring the ability to detect more subtle vegetation changes driven by fire. Historical aerial photographs offer a level of resolution between Landsat and on the ground plot data but have not been used yet to study patterns of chaparral cover change after wildfire. In this study, I measured the difference in vegetation cover following a long or short interval wildfire using historical aerial photographs. This approach allowed me to evaluate vegetation regrowth after numerous historical wildfires (1956-2003) with six to thirty-eight fire-free years postfire. In contrast to Meng et al. (2014), this method allowed for examination of the landscape at a much finer (one-meter) spatial resolution and over a longer time interval. Through evaluation of pre- and postfire images, I asked 1) does mature chaparral cover decline following a single short interval wildfire event in comparison to a single wildfire? and if so 2) where are chaparral communities declining? Furthermore, I explored 3) what landscape variables may help predict future chaparral vegetation losses or gains.

## II. METHODS

*A. Mapping the occurrence of short interval wildfires*

Fire history data (1879-2009) were acquired from the Fire and Resources Assessment Program (FRAP) database (CALFIRE, www.fire.ca.gov), reporting fires ≥ 4 hectares. The shapefile was then clipped to select for wildfires that fell within the study area of Ventura and Los Angeles Counties.

To map the occurrence of short interval wildfires, the fire history data were manipulated in ArcMap (ArcGIS 10.1) using the Feature to Polygon tool to create polygons with unique wildfire histories (e.g., “Fire Alarm Date”). A centroid point was then placed within each polygon with the Feature to Point tool and the two layers (the polygon layer and the point layer) were merged to count how many polygons overlapped each centroid point, following the “spaghetti and meatballs” technique (Honeycutt 2012, Moritz 2003). The output file was a merged shapefile with unique wildfire histories that preserved all the original wildfire perimeter information.

Next the attribute table of the merged shapefile was exported to Microsoft Excel (Microsoft) and new metrics such as “Minimum Fire Interval” and “Number of Fires” were calculated. Wildfire perimeters were corrected to eliminate single wildfires that were reported by multiple agencies, for example if a polygon had multiple “Fire Alarm

Dates" in the same year (Jacobson et al. 2004). The modified Excel table was finally joined back to the merged shapefile in ArcMap. The resulting polygons of the merged shapefile reflected the complete fire history of each location with original fire perimeters and Fire Alarm Dates from the fire history data as well as the calculated interval time (in years) that occurred between each wildfire at each location (Figure 1a, Table 1).

*B. Selection of twelve paired sites in Ventura and Los Angeles Counties*

Ventura and Los Angeles Counties are ideal for determining the effects of a short interval wildfire because the region is highly vulnerable to fire during the dry months (July-October) when Santa Ana wind conditions promote fast spreading wildfires (Hughes and Hall 2010). In addition, the number of short interval wildfires is predicted to increase as the population of southern California continues to grow (Keeley and Fotheringham 2001, Myers and Pitkin 2013).

To quantify the effect of a single short interval wildfire on vegetation in this region, twelve sites of adjacently paired polygons that experienced either one wildfire or two wildfires within the same five-year period were identified. Polygons that experienced two wildfires within five years were considered the "short interval fire" samples. Polygons that experienced one wildfire within the same five-year period were considered the "historical interval fire" samples; these "historical" polygons had, on average, experienced a wildfire 28.6 years prior to this study and two of the twelve polygons, at Site 004 and Site 110, had no prior record of fire since the early 1900s (Appendix A).

Beyond burn history, all polygons were 0.5 $km^2$ (50 hectares) or larger (Appendix A) and were selected along a moisture gradient from inland to the coast (Figure 1b). All sites were selected for locality in relation to the Santa Ana wind corridors. Areas within the Santa Ana wind corridors are more likely to experience multiple wildfires and thus a short interval wildfire, compared to areas outside the Santa Ana wind corridors.

Polygons that experienced both the first and second wildfire were labeled “twice burn”, where as “once burn” polygons only experienced the second wildfire. This choice was designed to capture the same number of regrowth years. For two of the twelve sites (Site 006 and 103), “once burn” polygons were selected from the first wildfire year which occurred ≥19 years prior to analysis. This exception was allowed assuming any difference in vegetation cover between once burn and twice burn polygons would be negligible after ≥19 total years of regrowth (Zammit and Zedler 1992).

*C. Selecting aerial photographs*

Historical aerial photographs (HAPs) were acquired from the Map and Imagery Laboratory (MIL) at the University of California Santa Barbara (www.library.ucsb.edu/mil). Prefire HAPs were selected as close to before the first wildfire as possible to record initial vegetation cover and postfire HAPs were selected six years or more following the second wildfire to capture maximal vegetation cover without encountering a third wildfire (Appendix A). Vegetation communities were assumed to return to prefire canopy cover within six years following wildfire (Muller et al. 1968, Schlesinger and Gill 1978). Seasonality of images was not controlled for under the assumption that mature communities appear distinguishable year-round (i.e., sage scrub communities reflect more light than evergreen chaparral communities and less light than alien annual grasses). Final HAP selection was based on availability for specific locations and timeframes (Table 2). HAPs were chosen between 1952 and 2009 for corresponding wildfires spanning 1956 to 2003.

*D. Georectifying aerial photographs*

To compare pre- and postfire vegetation cover on a pixel-by-pixel basis, all HAPs were georectified to the same base image. Grayscale, 2009, one-meter spatial resolution, digital orthophoto quarter quads (DOQQ) of Ventura or Los Angeles County, collected by the United States Geological Survey, were used as the base image. Temporally stable objects such as large shrubs or trees, rock outcrops, and crests and troughs of the mountainous landscape were used as registration points (RPs), observed at 12 times magnification. Dirt roads and permanent structures were also used, although these more permanent features were rare in the HAPs particularly in remote locations. The terrain of the HAPs was mountainous and highly variable so RPs were placed at a high density to increase warping accuracy. Each HAP was then warped using triangulation and pixels were resampled to the nearest neighbor, creating a georectified HAP with one-meter spatial resolution.

Georectified HAPs (gHAPs) were then mosaicked together to minimize edge distortion and to increase spatial accuracy for vegetation analysis. Mosaicked gHAPs covered the entire once burn and twice burn polygon of a site under prefire and postfire conditions. Only two sites were not georectified across their entire once burn polygon due to a lack of available HAPs and/or their extensive size. For Site 003, the entire twice burn polygon was georectified and an equivalent area within the once burn polygon was georectified. For Site 103, the entire twice burn polygon was georectified and approximately four times its area was georectified in the once burn polygon.

Mosaicked gHAPs were validated for their spatial accuracy by identifying 40-100 RPs corresponding to the 2009 DOQQ base map. Validation RPs had a final root mean square error of ten pixels (i.e., ten meters) or less.

*E. Subsite selection on north and south aspects*

Random points were generated in the prefire mosaicked gHAPs to select subsites for vegetation cover analysis. Each subsite was 50 x 50 pixels (50 x 50 meters) and was located in a once burn or twice burn polygon on either a north or south aspect (north: 0.0° to 67.5° or 292.5° to 360°; south: 112.5° to 247.5°) to account for differences in solar irradiance (northern aspects receive less solar irradiance than southern aspects) and soil moisture (Miller et al. 1983).

Thirteen to 20 subsites were randomly selected within each site for a total of 198 subsites (Appendix A). Ninety-nine subsites were located in once burn polygons and 99 subsites were located in twice burn polygons with 104 subsites on north facing aspects and 94 subsites on south facing aspects. Subsites were considered "independent" after including site as a covariate and finding no significant influence on subsite data.

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

Prescribed burns were reviewed and two of the 198 subsites overlapped with a prescribed burn. These two subsites were not omitted from analysis assuming they would not significantly change the trends found among subsites or sites.

*F. Quantifying vegetation cover within subsites*

To quantify vegetation cover at each subsite, the “dot grid” method was used (Floyd and Anderson 1982, Dublin 1991). A 10 x 10 grid (100 points) was overlaid on each subsite with a spacing of five pixels (five meters) between each point. Vegetation cover was observed at five times magnification and classified to life form: chaparral, alien annual grass, sage scrub, tree, or bare ground/exposed rock. All grass cover was assumed to be non-native based on the 1930’s Wieslander Maps and the 2001 USDA California Vegetation map. For classification consistency, all sites were examined twice to account for initial training and improvement in classification over time.

To improve classification accuracy, solar zenith was considered to account for shadows and Google Earth was referenced for cover and seasonal changes (available years: 1990-2015). Verification trips to the field were also conducted at six of the twelve sites. Current day photographs were taken from points of public access to capture vegetation cover of general areas within the fire perimeter and were compared against the HAPs for vegetation confirmation.

Vegetation cover was tallied to quantify total percent (%) vegetation cover by class at each subsite (100 points = 100% cover). Prefire vegetation cover (Figure 2, Appendix C.10) was subtracted from postfire vegetation cover to quantify the amount of vegetation change (“delta vegetation”) at each subsite. The impact of two wildfires in five years compared to one wildfire in five years was examined by comparing delta vegetation values for each vegetation class.

*G. Datasets for abiotic variables*

Aspect was calculated from USGS digital elevation models (DEMs) with a 30 x 30 meter horizontal resolution and a one-meter vertical resolution. Site location “outside” or “within” the Santa Ana wind corridors (Table 2) were visually determined from existing maps (Moritz et al. 2010) and from the Minimum Fire Interval map (Figure 1a). Distance from the coast was calculated in ArcMap by determining the centroid point of each polygon and measuring the shortest distance to the coastline “as the crow flies” (Table 2).

*H. Statistical analysis*

Statistics were calculated using RStudio (RStudio, Inc. version 0.98.1103) and were either run at the subsite level (e.g., 99 once burn subsites and 99 twice burn subsites) or at the polygon level (e.g., 12 once burn polygons and 12 twice burn polygons). Polygon values were calculated as the mean of subsite values.

A negative binomial regression was used to calculate the amount of change for each vegetation class by burn history (once burn: n = 99 subsites, twice burn: n = 99subsites), by burn history within a site (e.g., Site 113 – once burn: n = 8 subsites, twice burn: n = 8 subsites), by aspect (north: n = 104 subsites, south: n = 94 subsites) or by location in relation to the Santa Ana wind corridors (outside: n = 66 subsites, within: n = 132 subsites). An ANCOVA was used to calculate vegetation change across polygons by distance from the coast (once burn: n = 12 polygons, twice burn: n = 12 polygons) and time since fire (once burn: n = 12 polygons, twice burn: n = 12 polygons). Linear regressions (e.g., postfire cover by prefire cover for once burn and twice burn subsites) were performed in JMP12 (SAS, 2015).

## III. RESULTS

### *A. Average vegetation change by fire interval*

Subsite analysis (n = 198) showed no significant difference in delta values, i.e., percent (%) vegetation change from pre- to postfire between once burn and twice burn subsites, for all five vegetation classes (Figure 3, Appendix C.1). Average delta chaparral values were negative following either amount of wildfire (once burn = -0.76%, twice burn = -1.17%), although standard error bars for both wildfire histories approached or crossed zero ("no change"). The average pre- to postfire vegetation change for alien annual grass also showed no significant difference between once and twice burn subsites (0.37% and -0.02% respectively) and also had standard error bars that crossed zero ("no change"). The average delta sage scrub value in twice burn subsites was positive (2.28%), with standard error bars that did not cross zero, however it was not significantly different than the delta values in once burn subsites (0.05%, $p = 0.16$). Average delta tree and average delta bare ground values were negative in twice burn subsites and they were also not significantly different than once burn subsites.

When site was included as a covariate with fire history (once burn or twice burn), eleven of the twelve sites had similar delta values for each vegetation class. Site 2 was the only site that had a significantly more negative delta chaparral value and a significantly more positive delta alien annual grass value when a negative binomial regression was applied (see Appendix B for site details).

*B. Average vegetation change by location*

**Aspect: north or south**

When subsites were analyzed by north (n = 104) or south (n = 94) aspect, prefire conditions revealed northern aspects were typically dominated by chaparral and southern aspects were typically dominated by sage scrub (Figure 2, Appendix C.10). When calculated for percent (%) vegetation change, regardless of burn history, average delta chaparral and average delta sage scrub values showed significant differences (Figure 4, Appendix C.2). Delta chaparral values were significantly more negative on north aspects (-2.95%) compared to south aspects (1.23%) (negative binomial regression: $p = 0.010$). Delta sage scrub values were significantly more positive on north aspects (3.26%) compared to south aspects (-1.15%) (negative binomial regression: $p = 0.005$). No significant differences were found between north and south aspect for average delta alien annual grass, average delta tree, or average delta bare ground values (Figure 4, Appendix C.2). Subsites were further analyzed to include burn history and no significant differences were found in average delta vegetation values between once and twice burn sites on north facing or south aspects (Figure 5, Appendix C.3).

When site was included as a covariate with aspect, two of the twelve sites showed significantly different delta vegetation values. Site 2 had a more negative delta chaparral value and a more positive delta alien annual grass value and Site 104 had a more negative delta sage scrub value compared to the ten other sites when a negative binomial regression was applied (see Appendix B for site details).

## Santa Ana wind corridors: within or outside

The response of vegetation communities to wildfire, independent of wildfire history (once burn or twice burn), did not differ between subsites located within (n = 66) and outside (n = 132) of the Santa Ana wind corridors (Figure 6, Appendix C.4). Delta chaparral values were more negative outside the wind corridors (-1.43%) although they were not significantly different than delta chaparral values within them (0.02%) (negative binomial regression: $p = 0.42$). Similarly, average delta sage scrub values were more positive outside of the wind corridors (1.90%) but not significantly different than delta sage scrub values within them (0.30%) (negative binomial regression: $p = 0.19$). Analysis at the site level also showed no difference in delta values between sites located within the Santa Ana Corridors (n = 8) compared to outside of them (n = 16) (Appendix C.4).

Wildfire interval (once burn or twice burn) was included as a covariate along with location relative to the Santa Ana wind corridors and there were no significant differences in subsite delta values for all five vegetation classes (negative binomial regression, Figure 7, Appendix C.5). When site was included as a covariate, eleven of the twelve sites were similar in their overall response to wildfire. Site 2, located outside the Santa Ana wind corridors, had a significantly more negative delta chaparral value ($p = 0.0002$) and a significantly more positive delta alien annual grass value ($p = 0.0389$) compared to the eleven other sites (see Appendix B for site details).

Subsite location in relation to the Santa Ana wind corridors was further analyzed to include north and south aspects (Figure 8, Appendix C.6). Delta chaparral and delta sage scrub values again showed strong differences based on aspect regardless of location relative to the Santa Ana wind corridors. Delta chaparral values were strongly more negative on north aspects both outside (-3.39%, $p = 0.0453$) and within (-2.03%, $p = 0.0995$) the Santa Ana wind corridors and delta sage scrub values were significantly more positive on north aspects both outside (3.80%, $p = 0.0428$) and within (2.15%, $p = 0.0432$) of the Santa Ana wind corridors.

**Distance from coast**

A site's distance from the coast was not correlated with average delta values for any of the five vegetation classes whether analyzed by fire history (once burn: $n = 12$ polygons, twice burn: $n = 12$ polygons) or by overall trends ($n = 24$ polygons) (Appendix C.7). Even when delta values were standardized by the number of postfire years no strong correlation was observed (Appendix C.7).

*C. Average site vegetation change by time since fire*

Average delta values for chaparral and sage scrub cover were significantly correlated with time since fire (Appendix C.8, $n = 24$ polygons). Average delta chaparral values increased significantly with additional postfire years ($p = 0.0103$, slope = 0.2848) and average delta sage scrub values decreased significantly with additional postfire years ($p = 0.0145$, slope = -0.2662) independent of whether polygons were once or twice burned.

When examined by wildfire interval, once burn polygons were revealed to be driving the vegetation trends observed in postfire years (Figure 9, Appendix C.8). Average delta chaparral values in once burn polygons significantly increased with time since fire ($p = 0.01$, $R^2 = 0.50$, slope = 0.38) and average delta sage scrub values strongly decreased with time since fire ($p = 0.05$, $R^2 = 0.33$, slope = -0.28). In comparison, average delta chaparral and average delta sage scrub cover in twice burn polygons showed no significant correlation with number of postfire years ($p = 0.29$ and $p = 0.13$ respectively). Average delta alien annual grass, average delta tree, and average delta bare ground values showed no strong correlation with number of postfire years.

*D. Postfire by prefire correlation using subsite values*

Postfire values for chaparral, alien annual grass, sage scrub, and tree cover could be readily predicted by their prefire values (Figure 10, Appendix C.9, >90% of variation explained). However, there was a wide amount of variation observed among subsites (Figure 11). For example, Site 2, with six years of postfire recovery, had subsites with less chaparral recovery (negative delta values) or more alien annual grass recovery (positive delta values) than predicted by the 1:1 (prefire:postfire) regression. While Site 113, with twenty-two years of postfire recovery, had a subsite that increased in chaparral cover (positive delta value) more than the 1:1 (prefire:postfire) regression line.

Site 2 had the subsite with the largest increase in sage scrub cover from 0% prefire to 62% postfire, as well as three additional subsites that experienced large increases in sage scrub cover (0% prefire to 24%, 34%, or 45% postfire). Site 113 had the subsite with the largest decrease in sage scrub cover changing from 54% prefire to 8% postfire.

Regarding tree cover, Site 111 had one twice burn subsite that experienced a large decline in tree cover (77% prefire, 33% postfire) (Appendix C.9) following a two-year fire interval and twelve years postfire. Bare ground/exposed rock cover had relatively low prefire and postfire cover values and the regressions only explained 39% of once burn cover and 61% of twice burn cover.

## IV. DISCUSSION

In this study, a single short interval fire was not a significant factor in predicting vegetation change at the landscape scale (all subsites combined, Figure 3) nor did it result in a rapid increase of alien annual grass cover. Instead, all vegetation types were resilient to wildfire in terms of total cover, over the timescales measured (Figure 10, Appendix C.9). Despite this resilience, aspect (Figure 4) and time since wildfire (Figure 9) correlated with significant differences of chaparral and sage scrub cover between pre- and postfire images.

### *A. Vegetation change at the landscape level*

Prominent increases in alien annual grass cover following a single short interval wildfire were not observed across the landscape (Figure 3). Yet, while no differences were found at the landscape scale, changes were observed at the site, polygon, (Appendix B) and subsite scale (Figure 11). For example, there was one subsite where alien annual grass cover increased by 41% (six years postfire) and another that decreased by 45% (eight years postfire) compared to the prefire images. These findings suggest alien annual grass cover can change at the local subsite scale but assessment of it may be masked by analyses at the landscape scale. This also suggests that to better understand where vegetation change might be occurring, detailed images with higher resolution would be required for analysis.

Average sage scrub cover increased following a single short interval fire (Figure 3), which was similarly found by Jacobson et al. (2004). This suggests that a single short interval wildfire may drive conversion from a chaparral-dominated community to a sage scrub-dominated community. In other words, sage scrub may be an intermediate step towards loss of chaparral. Perhaps repeat short interval fires at a single location are needed to cause a conversion from chaparral all the way to alien annual grasses, as has been shown in models where ongoing wildfires occur (Syphard 2006).

Chaparral communities decreased an average of only 1% following any amount of wildfire, although the change was not different than "no change" (Figure 3, Appendix C.1). These data suggest that chaparral communities are resilient to wildfires at the scale measured or that other factors, such as postfire precipitation or fire intensity (Keeley et al. 2005b), are more important to chaparral recovery than just a single short wildfire interval alone.

Another explanation as to why significant differences were not found between chaparral cover at once burn and twice burn polygons is that the HAPs used in this study were not of high enough resolution (one-meter resolution) to detect species compositional change within the shrubland communities. We could readily differentiate between vegetation life forms (e.g., grasses, deciduous shrubs, chaparral shrubs) but not plant species. This is important because chaparral species have multiple ways of recruiting or recovering after wildfire including species classified as obligate seeders, obligate resprouters, or facultative resprouters (Keeley 1991, Syphard et al. 2006). Short interval wildfire events could be selecting against certain species or types of species, but we were not able to detect this.

Obligate seeders only recruit from the surviving seedbank and all adult shrubs are consumed by the fire (e.g., *Ceanothus megacarpus, Ceanothus gregii, Arctostaphylos glauca*). These species require seven to fifteen years to reach reproductive maturity (Zammit and Zedler 1992) and are at risk of extirpation if another fire comes through before the seedbank has been replenished (Zedler et al. 1983, Keeley and Brennan 2012). Obligate resprouters, on the other hand, only recruit from the lignotuber (the underground carbon stores) of a surviving adult (e.g., *Heteromeles arbutifolia, Quercus berberidifolia, Cercocarpus betuloides*) as their seeds are killed by fire (Keeley 1991). Finally, facultative resprouters can recruit from the seedbank as well as from surviving adults (e.g., *Adenostoma fasciculatum, Malosma laurina, Arctostaphylos glandulosa, Rhus ovata*).

The ability to quickly resprout following fire is advantageous as shrubs can readily acquire resources (i.e., sun, water, nutrients) in a low competition environment. Indeed, facultative resprouters, such as *M. laurina*, have been known to have higher resprout survivorship than obligate seeders under the same postfire conditions (Thomas and Davis 1989). These different recruitment strategies are important because they could explain why some delta values for chaparral are similar between "historical" and short interval wildfires. If chaparral species with resprouting capabilities are strongly competitive they could outcompete obligate seeders (Tyler and D'Antonio 1995) causing a mixed chaparral stand to become a monoculture of *A. fasciculatum* or *M. laurina*, for example. While recruitment strategy is an important component to consider, this study used one-meter HAPs to evaluate landscape-scale changes and not compositional changes.

*B. Vegetation change due to aspect*

Chaparral cover decreased (-3.0%) on north aspects and unexpectedly increased (1.2%) on south aspects (Figure 4). An increase on south aspects following wildfire was unexpected because south aspects tend to have more xeric conditions, receiving more solar radiation and less water for long-lived shrubs to thrive (Miller et al. 1983). One explanation of why chaparral communities could have increased in cover on south aspects is due to the expansion of *M. laurina*, a tenacious facultative resprouter as discussed above, which can outcompete other species when water resources are limited (Thomas and Davis 1989). Because these shrubs can reach similar heights and canopy densities as mixed chaparral stands, monocultures of *M. laurina* would have been classified as "chaparral" in the HAPs.

An explanation as to why chaparral cover did not increase on north aspects could be due to the method of data collection. Since values from the dot-grid method only span 0-100%, subsites with close to 100% chaparral cover prefire had little room to increase in chaparral cover postfire. At subsites that had 100% chaparral cover prefire, only a decrease in cover or "no change" was possible. Indeed, average prefire chaparral cover on north aspects was 72% compared to only 26% on south aspects (Figure 2, Appendix C.10). Standardizing the data in relation to prefire cover was attempted but calculations often inflated non-biologically important results and hid larger community changes. For example, an increase from 1% to 3% cover would be a 3-fold change whereas an increase from 50% to 75% cover would only be a 1.5-fold change.

*C. Vegetation change due to time since fire*

Chaparral and sage scrub communities showed strong yet opposing trends in recovery following fire. Not accounting for fire history, chaparral communities overall increased significantly ($p = 0.01$) with additional years after fire whereas sage scrub cover strongly declined ($p = 0.05$) with additional years after fire (Appendix C.8, $n = 24$). When accounting for fire history, average delta chaparral values were negative until 17 or 22 years (once burn or twice burn respectively) and average delta sage scrub values were positive until 15 or 25 years (once burn or twice burn respectively) (Figure 9). This is in support of previous studies that suggest sage scrub is successional to chaparral and thus sage scrub cover dominates during the first 15 years postfire but slowly declines as individuals are shaded out by the canopy of chaparral shrubs (McPherson and Muller 1967, Gray 1983). This dynamic in canopy cover is concurrent with the assumption that chaparral communities require five to thirty years to recover from wildfire (Hanes 1971, Hope et al. 2007, Schlesinger and Gill 1978) and subshrubs, common in sage scrub communities, decline as chaparral shrubs mature around them.

The increase in chaparral cover, beyond prefire conditions, could be explained by a shift in species composition from, say, one species with a smaller canopy to a species with a more expansive canopy. This again illustrates the resolution limitation of the HAPs; vegetation life form can be identified at a 1 x 1 meter resolution although identifying species is not possible.

Another explanation to why chaparral cover exceeded prefire conditions could be due to site variation. Site 104 had 22 years of regrowth preceding the prefire HAPs and 38 years, the longest duration of regrowth, between the second wildfire and the site's postfire HAPs (Table 2). It was also the most inland site (Table 2) and had almost no south facing subsites: six out of eight once burn subsites and nine out of nine twice burn subsites were on northern aspects. The other eleven sites had approximately equal proportions of subsites with northern and southern aspects (e.g., four subsites on northern aspects and four subsites on southern aspects). However, when Site 104 was omitted, the rate of chaparral change over time increased from 0.38 to 0.47 for once burn subsites and from 0.19 to 0.25 for twice burn subsites (Table 3). The rates of chaparral change over time calculated with all 12 sites (12 once burn polygons and 12 twice burn polygons) are more consistent with the current thought that chaparral-dominated communities increase slowly over time if they increase at all (Hanes 1971, Keeley 1986). Presumably the rate of chaparral change would eventually reach an asymptote as the amount of chaparral cover at each subsite cannot exceed 100% cover.

Finally, an increase of 2% to 6.5 % in chaparral cover might actually be occurring over a postfire period of 38 years. As modeled in Syphard et. al (2006), other vegetation community types were predicted to convert to chaparral communities under the longest fire treatment (average fire return interval of 10 years). Thus, it could be possible that with enough fire-free years, chaparral communities could expand and other communities could convert to a chaparral-dominated community.

*D. Vegetation resilience to fire*

The strong correlation between prefire and postfire cover (Figure 10) suggests strong resilience for all vegetation types in response to wildfire disturbance at the classification scale of 'vegetation type'. Yet there was evidence for vegetation change at the local scale (Figure 11); some subsites had large increases or decreases in cover that did not fit the overall trend. Subsites at Site 2 for example, experienced large declines in chaparral cover (Figure 11a), which could be due to the site's minimal number of postfire years following fire (six years) or because it is the only site that experienced a previous short interval fire many years earlier (Appendix A). Site 113 also had subsites that experienced a decline in chaparral cover following fire (Figure 11b) and it is not clear what is different about this site compared to the others. Again, these results suggest that, even though landscape scale changes among communities were minimal, more dramatic changes can occur at the local scale.

*E. Additional wildfire history*

The goal of this study was to consider the effect of a single short interval wildfire on southern California vegetation communities. Results may have been different if sites which experienced repeated short interval wildfire were included or if the entire wildfire history (1878-2009) of a location was considered in the analysis. As discussed by Zedler (1995), the variability in years between fires may be equally important as the number of short interval fires that occur. Further investigation into sites that experienced more than one short interval wildfire would be valuable, though it becomes more challenging to identify comparable sites for analysis.

Another factor that might have led to different results is the time frame of analysis (i.e., 1956-2003). Perhaps resilient shrub species had already been selected for in locations that experienced multiple wildfires. In this situation, any vegetation change that would have occurred due to a short interval wildfire had already occurred, prior to 1956 when our first imagery was available. This could also explain why there was no observable difference in vegetation change between sites within and outside of the Santa Ana wind corridors (Figure 6, Appendix C.4). Perhaps vegetation communities within the Santa Ana wind corridors had already been selected for resilient species, resulting in almost no change among sites within the wind corridors and a greater amount of vegetation change outside of the wind corridors, where sites may be more vulnerable to wildfire due to limited exposure to previous wildfires.

*F. External factors and the future*

While this study did not find a consistent increase in alien annual grass cover following a single short interval wildfire in this region, these alien annual grasses likely still pose a risk to the surrounding vegetation communities. They create highly ignitable fuel across the landscape when they senesce and can carry wildfire into chaparral stands that otherwise would not have ignited (Keeley 2001). The Zaca Fire (2007), Jesusita Fire (2009), and Springs Fire (2013) are examples of fires that started in dry alien annual grass and quickly spread into chaparral.

An expanding wildland-urban-interface is expected to create additional risks for southern California's wildlands as new urban development pushes into existing shrublands. As Syphard et al. (2007a) found, ignition risk is highest with intermediate human populations (35-45 people/km$^2$) and an intermediate mix of urban development and wildland communities. Climate change will also pose an increased threat to wildlands as the climate in southern California becomes hotter and possibly drier (Krawchuk and Moritz 2012). Periods of drought are expected to lengthen and rainfall events are expected to become more punctuated. With extended periods of drought-like conditions, wildfire seasons will further expand as live fuel moisture diminishes and alien annual grasses will senesce earlier providing higher ignition risk throughout a longer period of the year.

## V. CONCLUSION

The aim of this study was to identify if mature vegetation communities obviously change to other vegetation types in response to a single short interval wildfire (defined as two wildfires within five years), and where changes in vegetation might be occurring on the landscape. These results showed chaparral cover declined following wildfire, regardless of fire history, and sage scrub cover, rather than alien annual grass cover, increased following a single short interval wildfire. We found little support for the assertion that a single short interval wildfire could convert chaparral to alien annual grassland.

In predicting vegetation change due to wildfire, aspect was the best predictor of differences in chaparral and sage scrub cover while time since fire was the best predictor of chaparral and sage scrub stand regrowth, especially following a “historical” wildfire interval. Postfire and prefire cover were highly correlated for all vegetation types, although variations among subsites were apparent, suggesting that vegetation change occurring at the local scale may not be statistically significant at the landscape scale.

In conclusion, vegetation community types of southern California are resilient but their resilience following wildfire is dependent on a multitude of factors. While overall vegetation types appeared to be relatively stable in response to wildfire regardless of the wildfire histories examined here, there may be specific locations that are more susceptible to change that were not detected under a landscape level analysis. Furthermore, caution is advised when managing these vegetation communities since little is still known about their actual successional trajectories or how environmental variables (e.g., amount of precipitation in postfire years) influence recovery at both the landscape scale and the field scale. Fire regime is only one factor that may lead to changes in vegetation and possibly stronger drivers are aspect, prefire and postfire conditions, and time since wildfire. These additional variables need to be considered when predicting changes in vegetation communities due to wildfires in southern California.

## VI. REFERENCES


Barrett, S. W., S. F. Arno. 1982. Indian fires as an ecological influence in the Northern Rockies. *Journal of Forestry* 80: 647-651.

Bentley, J. R. 1967. Conversion of chaparral areas to grassland: techniques used in California. Agriculture Handbook No. 328. U.S. Forest Service Department of Agriculture.

Brooks, M. L, C. M. D'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. DiTomaso, R. J. Hobbs, M. Pellant, D. Pyke. 2004. Effects of invasive alien plants on fire regimes. *BioScience* 54(7): 677-688.

D'Antonio, C. M. and P. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics* 23: 63-87.

Dublin, H. T. 1991. Dynamics of the Serengeti-Mara woodlands: an historical perspective. *Forest & Conservation History* 34(4): 169-178.

Everett, R. L., R. Schellhaas, D. Keenum, D. Spurbeck, P. Ohlson. 2000. Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. *Forest Ecology and Management* 129: 207-225.

Floyd, D.A. and J.E. Anderson. 1982. A new point interception frame for estimating cover of vegetation. *Plant Ecology* 50(3): 185-186.

Fuhrmann, K. N. and T. E. Crews. 2001. Long-term effects of vegetation treatments in the chaparral transition zone. *Rangelands* 23(1): 13-16.

Gray, J. T. 1983. Competition for light and a dynamic boundary between chaparral and coastal sage scrub. *Madroño* 30(1): 43-49.

Haidinger, T. L. and J. E. Keeley. 1993. Role of high fire frequency in destruction of mixed chaparral. *Madroño* 40(3): 141-147.

Hanes, T. L. 1971. Succession after fire in the chaparral of southern California. *Ecological Monographs* 41(1): 27-52.

Honeycutt, D. 2012. More adventures in overlay: counting overlapping polygons with spaghetti and meatballs. *ArcGIS Resources*. ESRI.

Hope, A., C. Tague, R. Clark. 2007. Characterizing post-fire vegetation recovery of California chaparral using TM/ETM+ time-series data. *International Journal of Remote Sensing* 28(6): 1339-1354.

Hughes, M and A. Hall. 2010. Local and synoptic mechanisms causing Southern California's Santa Ana winds. *Climate Dynamics*. 34: 847-857.

Jacobson, A. L., S. D. Davis, S. L. Fabritius. 2004. Fire frequency impacts non-sprouting chaparral shrubs in the Santa Monica Mountains of southern California. In: Arianoutsou, M and V. P. Papanastasis, eds. Ecology, Conservation and Management of Mediterranean Climate Ecosystems. Rotterdam, Netherlands: Millpress. 6 p.

Keeley, J. E. 1986. Resilience of mediterranean shrub communities to fires. In: Dell, B., A. J. M. Hopkins and B. B. Lamont, eds. Resilience in Mediterranean-type Ecosystems. Netherlands: Springer. 95-112 p.

Keeley, J. E. 1987. Role of fire in seed germination of woody taxa in California chaparral. *Ecology* 68(2): 434-443.

Keeley, J. E. 1991. Seed germination and life history syndromes in the California chaparral. *The Botanical Review* 57: 81-116.

Keeley, J. E. 2001. Fire and invasive species in Mediterranean-climate ecosystems of California. Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication No. 11: 81-94.

Keeley, J. E. and C. J. Fotheringham. 2001. Historic fire regime in southern California shrublands. *Conservation Biology* 15(6): 1536-1548.

Keeley, J. E., C. J. Fotheringham, M. A. Moritz. 2004. Lessons from the October 2003 wildfires in southern California. *Journal of Forestry* October/November 2004: 26-31.

Keeley, J. E., C. J. Fotheringham, M. Baer-Keeley. 2005a. Determinants of postfire recovery and succession in Mediterranean-climate shrublands in California. *Ecological Applications* 15: 1515-1534.

Keeley, J. E., C. J. Fotheringham, M. Baer-Keeley. 2005b. Alien plant dynamics following fire in Mediterranean-climate California shrublands. Ecological Applications 15(6): 2109-2125.

Keeley, J. E., C. J. Fotheringham, M. Baer-Keeley. 2006. Demographic patterns of postfire regeneration in Mediterranean-climate shrublands of California. *Ecological Monographs* 76(2): 235-255.

Keeley, J.E. and T.J. Brennan. 2012. Fire-driven alien invasion in a fire-adapted ecosystem. *Oecologia* 169: 1043-1052.

Krawchuck, M. A. and M. A. Moritz (Simon Fraser University; University of California, Berkeley). 2012. Fire and climate change in California. California Energy Commission. Publication number: CEC-500-2012-026.

McPherson, J. K. and C. H. Muller. 1967. Light competiton between *Ceanothus* and *Salvia* shrubs. *Bulletin of the Torrey Botanical Club* 94(1): 41-55.

Meng, R., P. E. Dennison, C. M. D'Antonio, M. A. Moritz. 2014. Remote sensing analysis of vegetation recovery following short-interval fires in southern California shrublands. *PloS ONE* 9(10): e110637.

Miller, P. C., D. K. Poole, P. M. Miller. 1983. The influence of annual precipitation, topography, and vegetation cover on soil moisture and summer drought in southern California. *Oecologia (Berlin)* 56: 385-391.

Moritz, M. A. 2003. Spatiotemporal analysis of controls on shrubland fire regimes: age dependency and fire hazard. *Ecology* 84(2): 351-361.

Moritz, M. A., T. J. Moody, M. A. Krawchuk, M. Hughes, A. Hall. 2010. Spatial variation in extreme winds predicts large wildfire locations in chaparral ecosystems. *Geophysical Research Letters* 37, L04801, doi:10.1029/2009GL041735.

Muller, C. H., R. B. Hanawalt, J. K. McPherson. 1968. Allelopathic control of her growth in the fire cycle of California chaparral. *Bulletin of the Torrey Botanical Club* 95(3): 225-231.

Myers, D. and J. Pitkin. 2013. The Generational Future of Los Angeles: Projections to 2030 and Comparisons to Recent Decades. Population Dynamics Research Group, Sol Price School of Public Policy, University of Southern California.

Nowacki, G. J. and M. D. Abrams. 2008. The demise of fire and "mesophication" of forests in the Eastern United States. *BioScience* 58(2): 123-138.

Peterson, S. H. and D. A. Stow. 2010. Using multiple image endmember spectral mixture analysis to study chaparral regrowth in southern California. *International Journal of Remote Sensing* 24(22): 4481-4504.

Peterson, S. H., M. A. Moritz, M. E. Morais, P. E. Dennison, J. M. Carlson. 2011. Modelling long-term fire regimes of southern California shrublands. *International Journal of Wildland Fire* 20:1-16.

Rosario, J. A. and E. W. Lathrop. 1974. Comparison of vegetation structure and composition in modified and natural chaparral. *Journal of Range Management* 27(4): 310-312.

Schlesinger, W. H and D. S. Gill. 1978. Demographic studies of the chaparral shrub, *Ceanothus megacarpus*, in the Santa Ynez Mountains, California. *Ecology* 59(6): 1256-1263.

Syphard, A. D., J. Franklin, J. E. Keeley. 2006. Simulating the effects of frequent fire on southern California coastal shrublands. *Ecological Applications* 16(5): 1744-1756.

Syphard, A. D., V. C. Radeloff, J. E. Keeley, T. J. Hawbaker, M. K. Clayton, S. I. Stewart, R. B. Hammer. 2007a. Human influence on California fire regimes. *Ecological Applications* 17(5): 1388-1402.

Syphard, A. D., K. C. Clarke, J. Franklin. 2007b. Simulating fire frequency and urban growth in southern California shrublands, USA. *Landscape Ecology* 22:431-445.

Thomas, C. M. and S. D. Davis. 1989. Recovery patterns of three chaparral shrub species after wildfire. *Oecologia* 80(3): 309-320.

Tyler, C. M. and C. M. D'Antonio. 1995. The effects of neighbors on the growth and survival of shrub seedlings following fire. *Oecologia* 102(1): 255-264.

Zammit, C. A. and P. H. Zedler. 1992. Size structure and seed production in even-aged populations of *Ceanothus greggii* in mixed chaparral. *Journal of Ecology* 81:499-511.

Zedler, P.H., C.R. Gautier, G.S. McMaster. 1983. Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. *Ecology* 64: 809-818.

Zedler, P. H. 1995. Fire frequency in southern California shrublands: Biological effects and management options. *Brushfires in California Wildlands: Ecology and Resource Management* pp. 101-112.

## VII. FIGURES

**Figure 1. (a) Minimum fire intervals as reported in the CalFire database (frap.fire.ca.gov) for Ventura and Los Angeles County from 1878 to 2009. (b) Twelve sites where one polygon burned twice within five years (“twice burn”, red) and the other polygon burned once within the same five-year period (“once burn”, orange).**

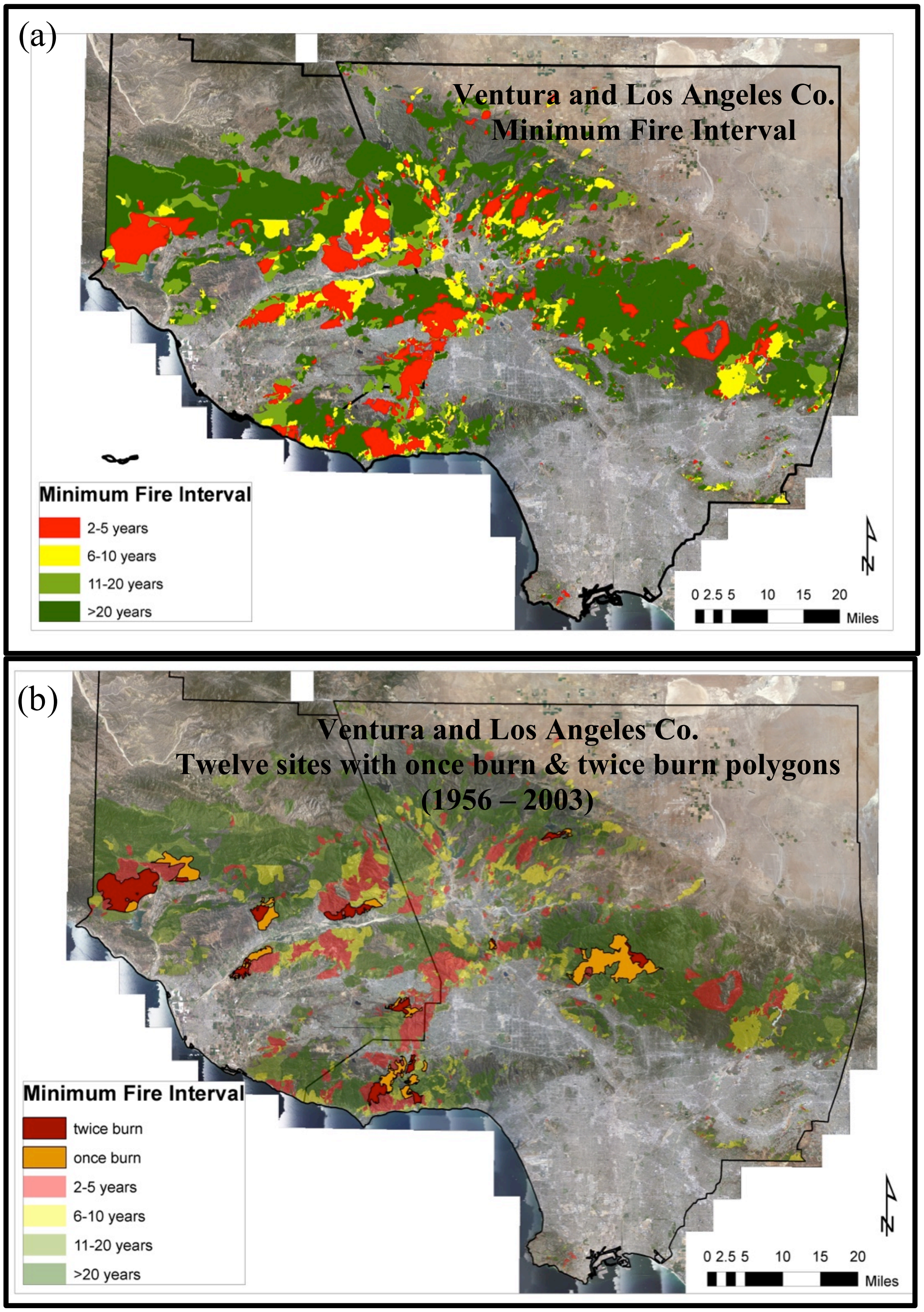


**Figure 2. Average prefire vegetation cover by burn history and aspect ± one standard error across 198 subsites. See Appendix B for site details.**

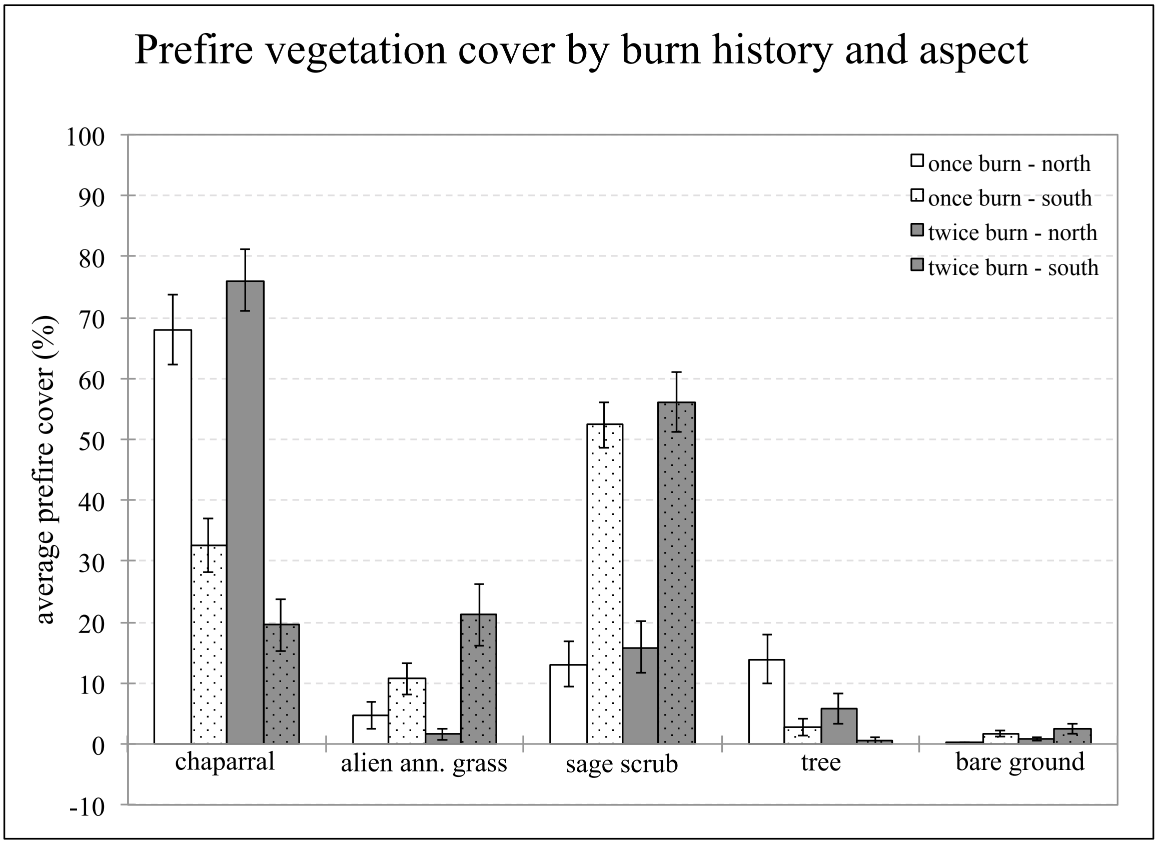


**Figure 3. Average vegetation change (%) following one or two fires in the same five-year period ± one standard error across 198 subsites. A value of 0 signifies "no change" between postfire and prefire conditions. Site was considered as a covariate in a negative binomial regression analysis and was not significant except for Site 2 (chaparral, alien annual grass). See Appendix B for site details.**

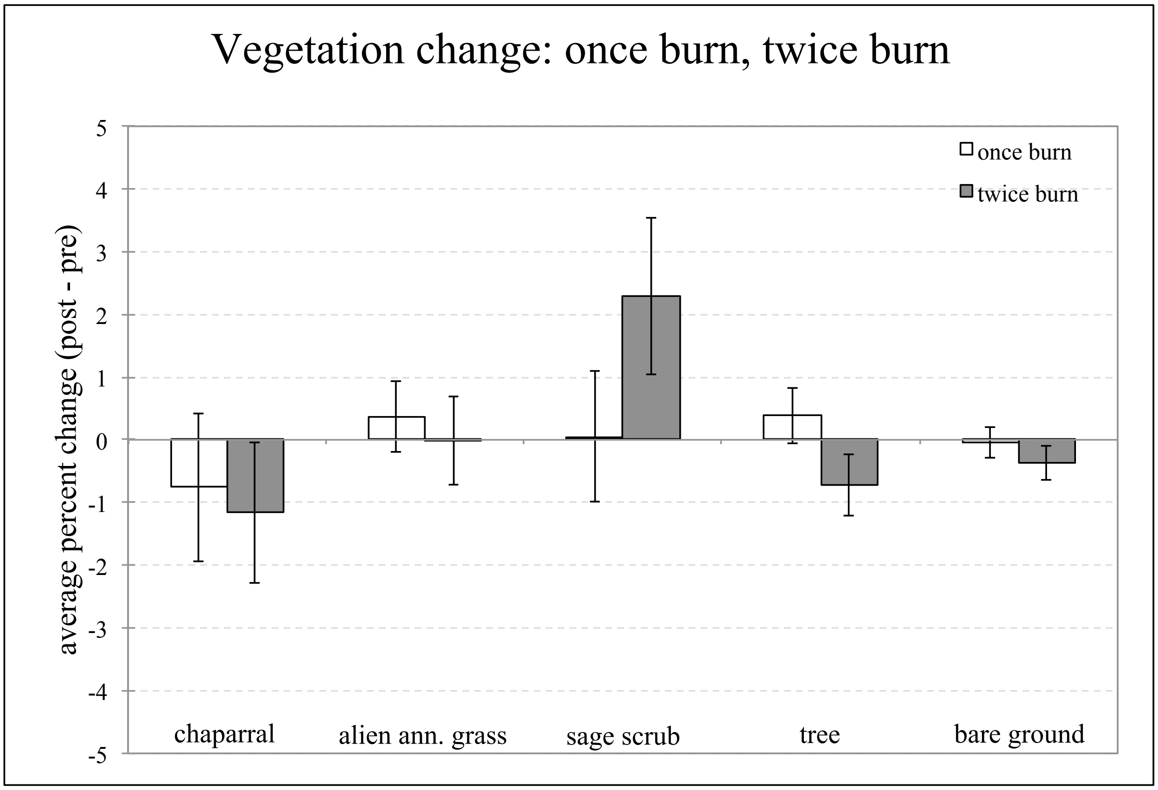


**Figure 4. Average vegetation change (%) on north or south aspect ± one standard error across 198 subsites. Site was considered as a covariate in a negative binomial regression analysis and was not significant except for Site 2 (chaparral, alien annual grass) and Site 104 (sage scrub) See Appendix B for site details. ** p-value < 0.01**

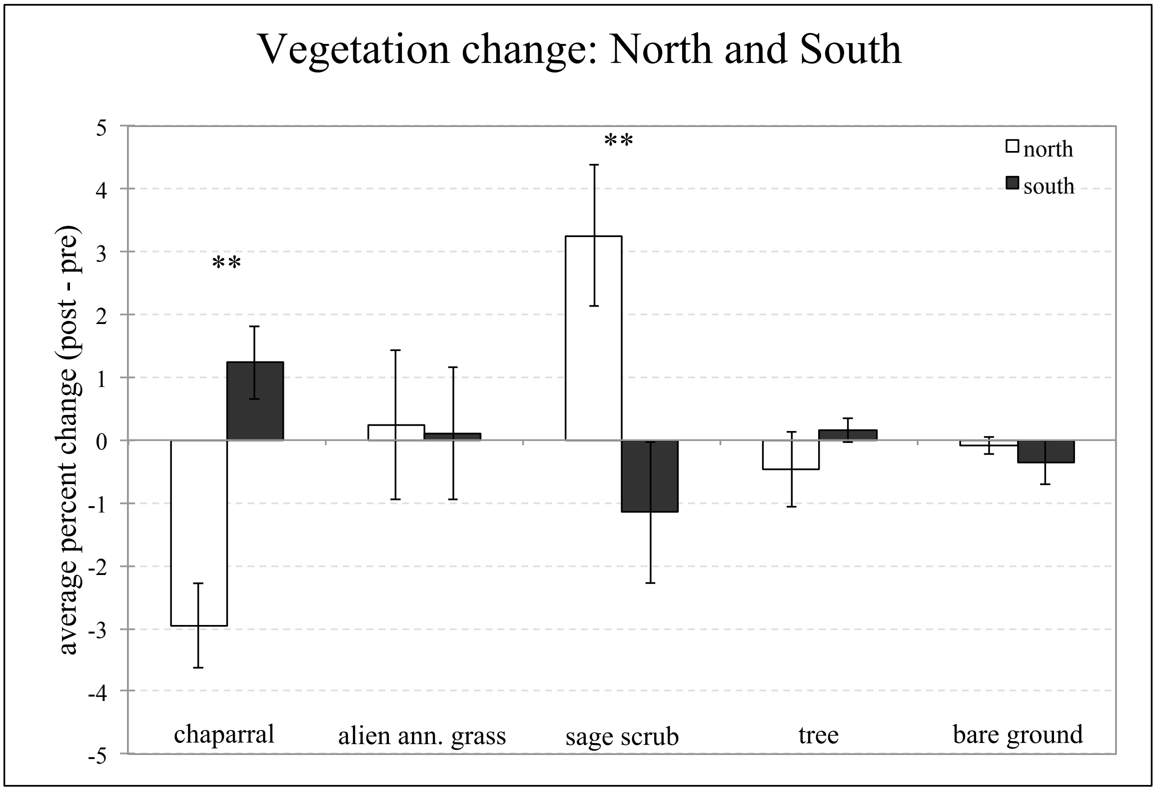


**Figure 5. Average vegetation change (%) by fire history on (a) north or (b) south aspect ± one standard error across 198 subsites. Significance determined by a negative binomial regression analysis. Site was considered as a covariate and was not significantly different except for Site 2 (chaparral, alien annual grass, sage scrub), 104 (sage scrub), and 109 (chaparral). See Appendix B for site details.**

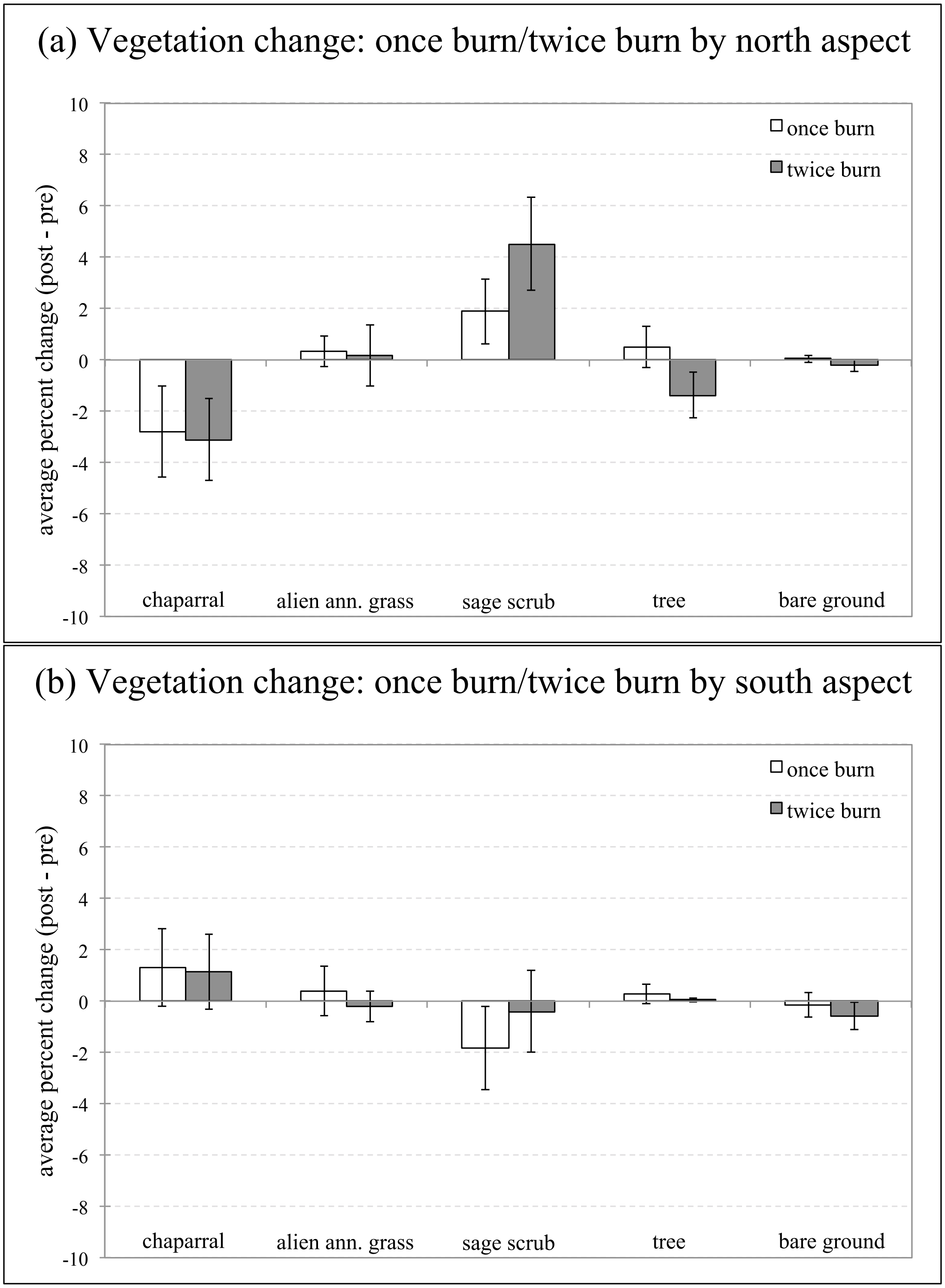


**Figure 6. Average vegetation change (%) located outside or within the Santa Ana wind corridors ± one standard error across 198 subsites. Site was considered as a covariate in a negative binomial regression analysis and was not significant except for Site 2 (chaparral, alien annual grass). See Appendix B for site details.**

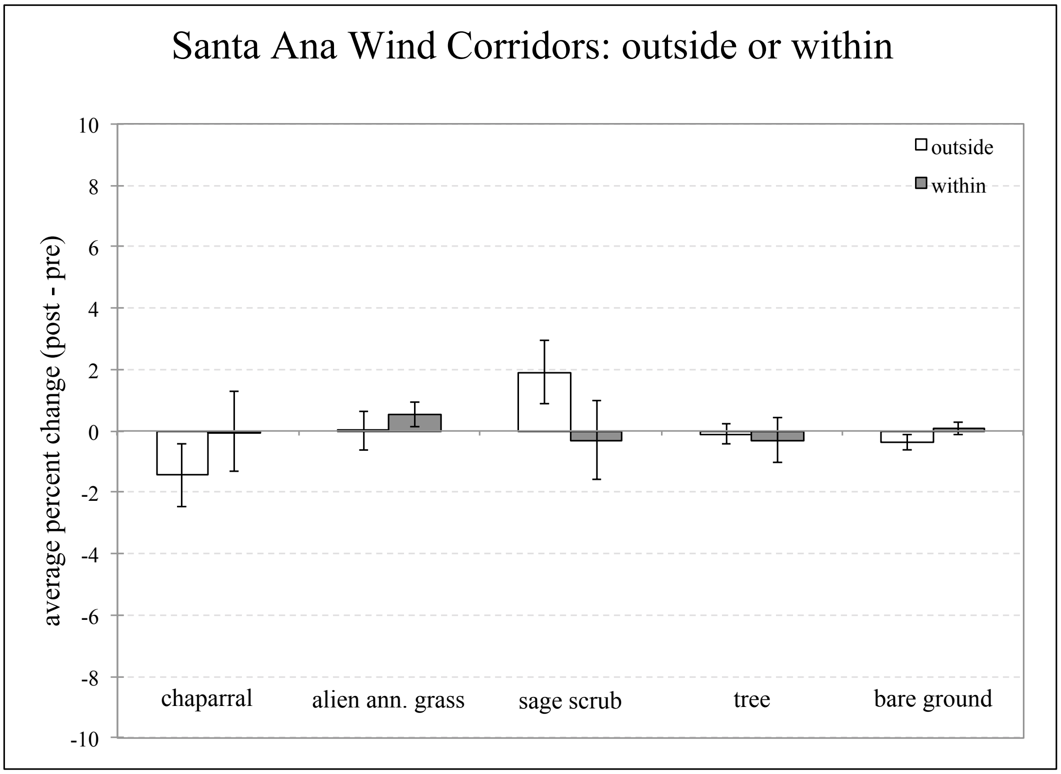


**Figure 7. Average vegetation change (%) by burn history located (a) outside or (b) within the Santa Ana wind corridors ± one standard error across 198 subsites. Site was considered as a covariate in a negative binomial regression analysis and was not significant except for Site 2 (chaparral, alien annual grass) outside the Santa Ana wind corridors. See Appendix B for site details.**

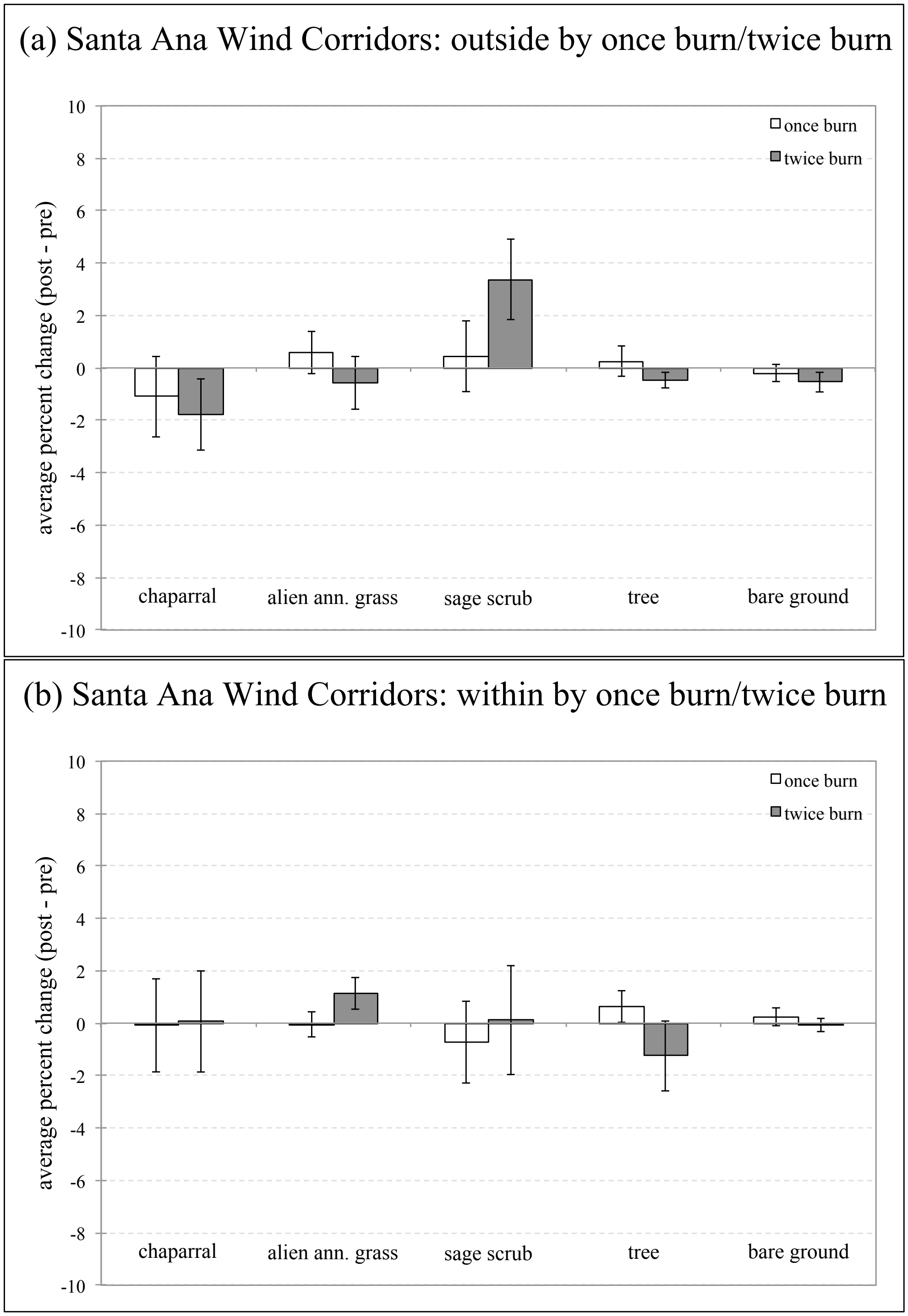


**Figure 8. Average vegetation change (%) by aspect located (a) outside or (b) within the Santa Ana wind corridors ± one standard error across 198 subsites. Site was considered as a covariate in a negative binomial regression analysis and was not significant except for Site 2 (chaparral, alien annual grass) and Site 104 (sage scrub) outside the Santa Ana wind corridors. See Appendix B for site details.**
*** p-value < 0.05, + p-value < 0.1**

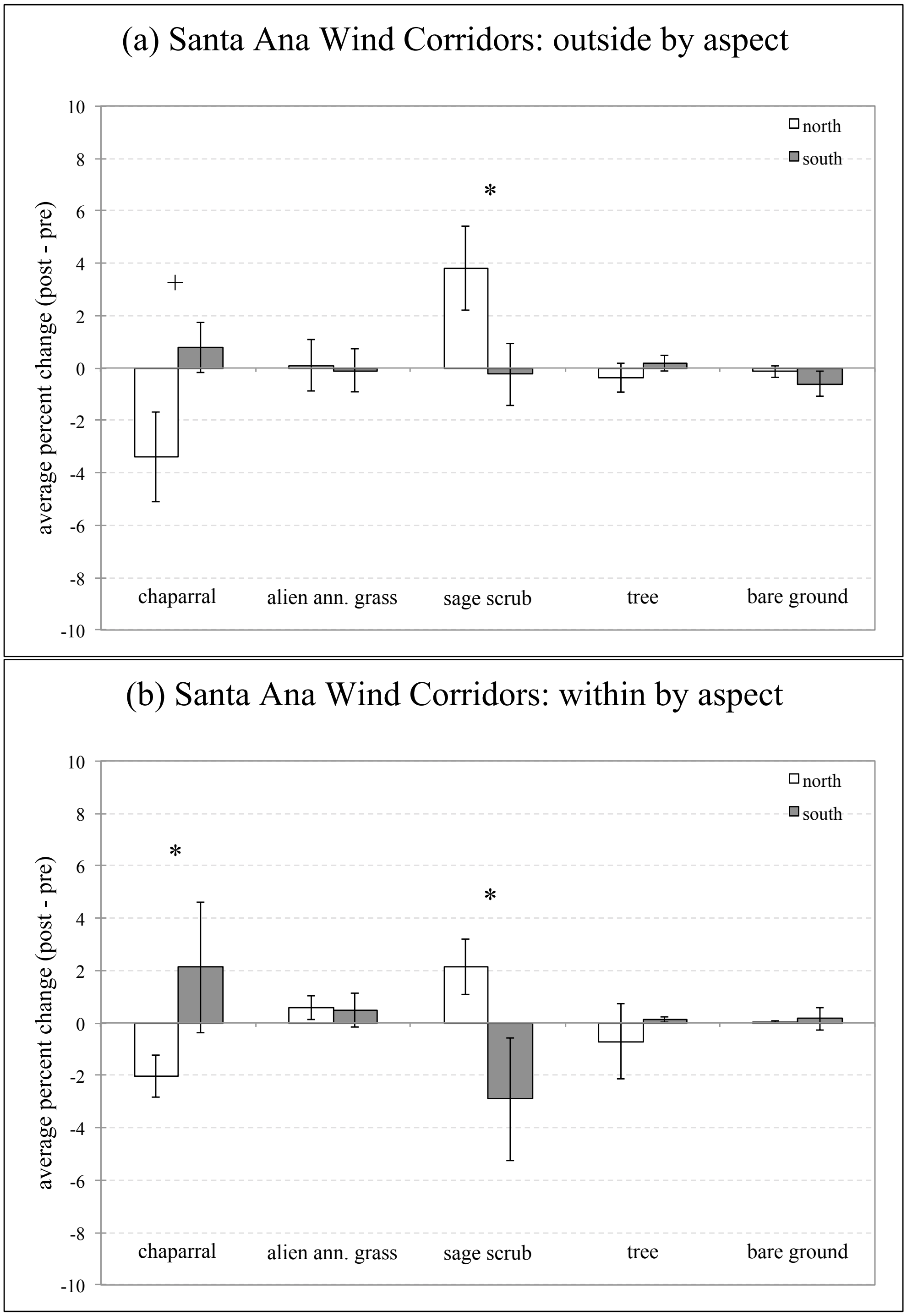


**Figure 9. Average vegetation change (%) by time since wildfire for once burn (n = 12) and twice burn (n = 12) polygons. "Years since fire" is equivalent to Years of Regrowth in Table 2.**

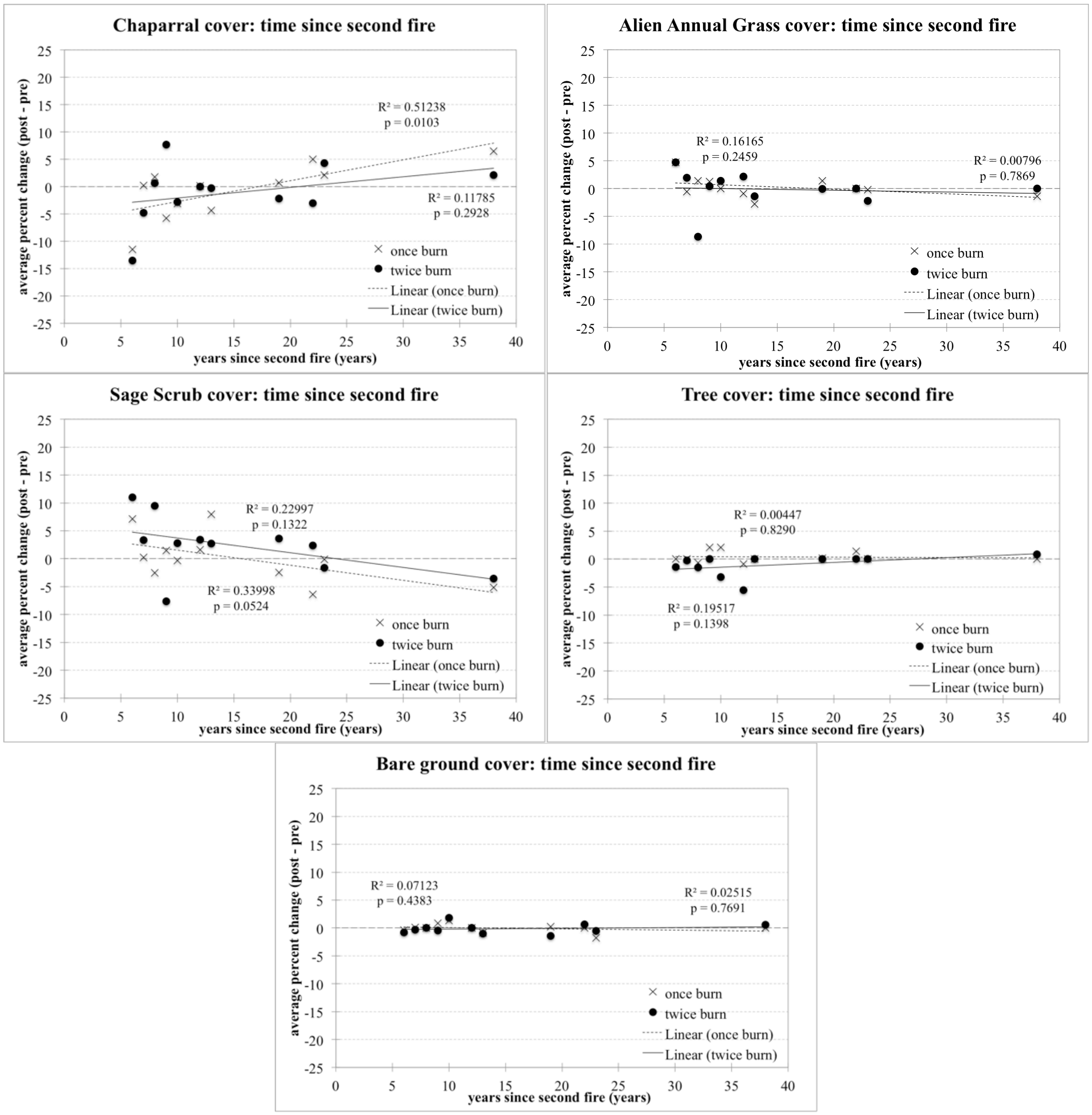
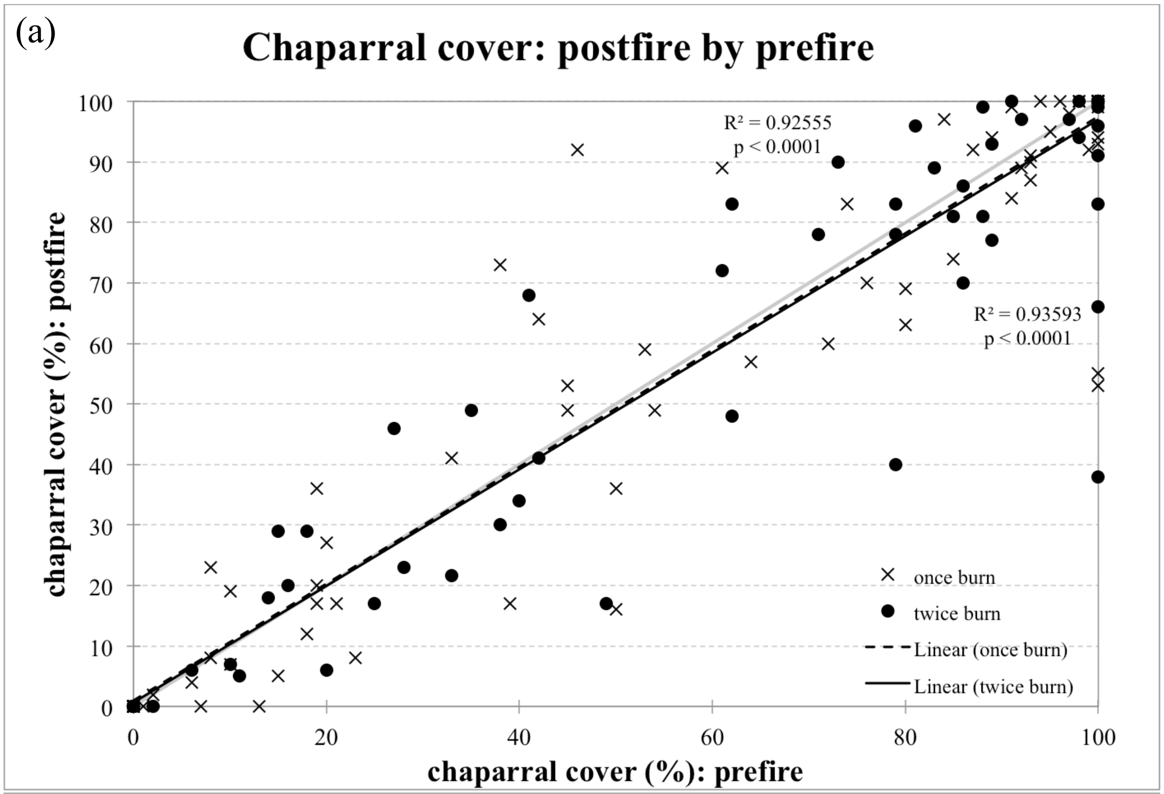


**Figure 10. Postfire by prefire vegetation cover (%) for (a) chaparral and (b) sage scrub cover considering 198 subsites. Gray line represents 1:1 ratio.**



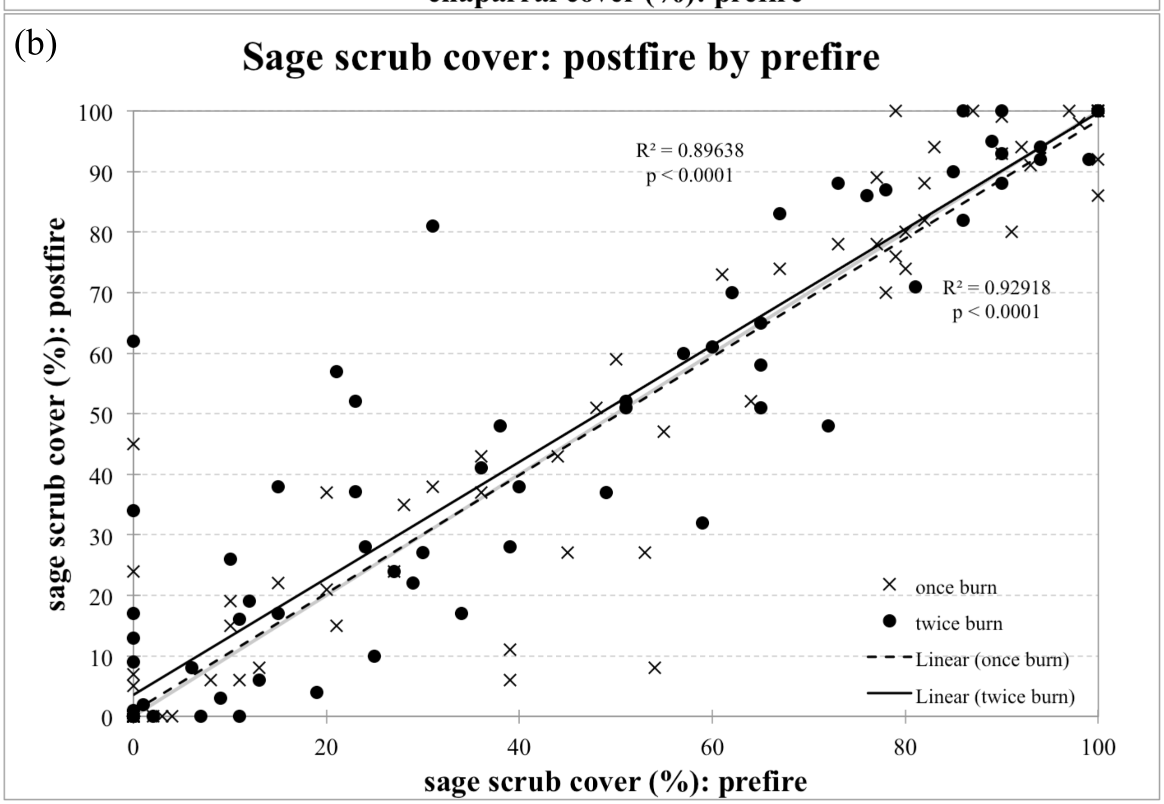
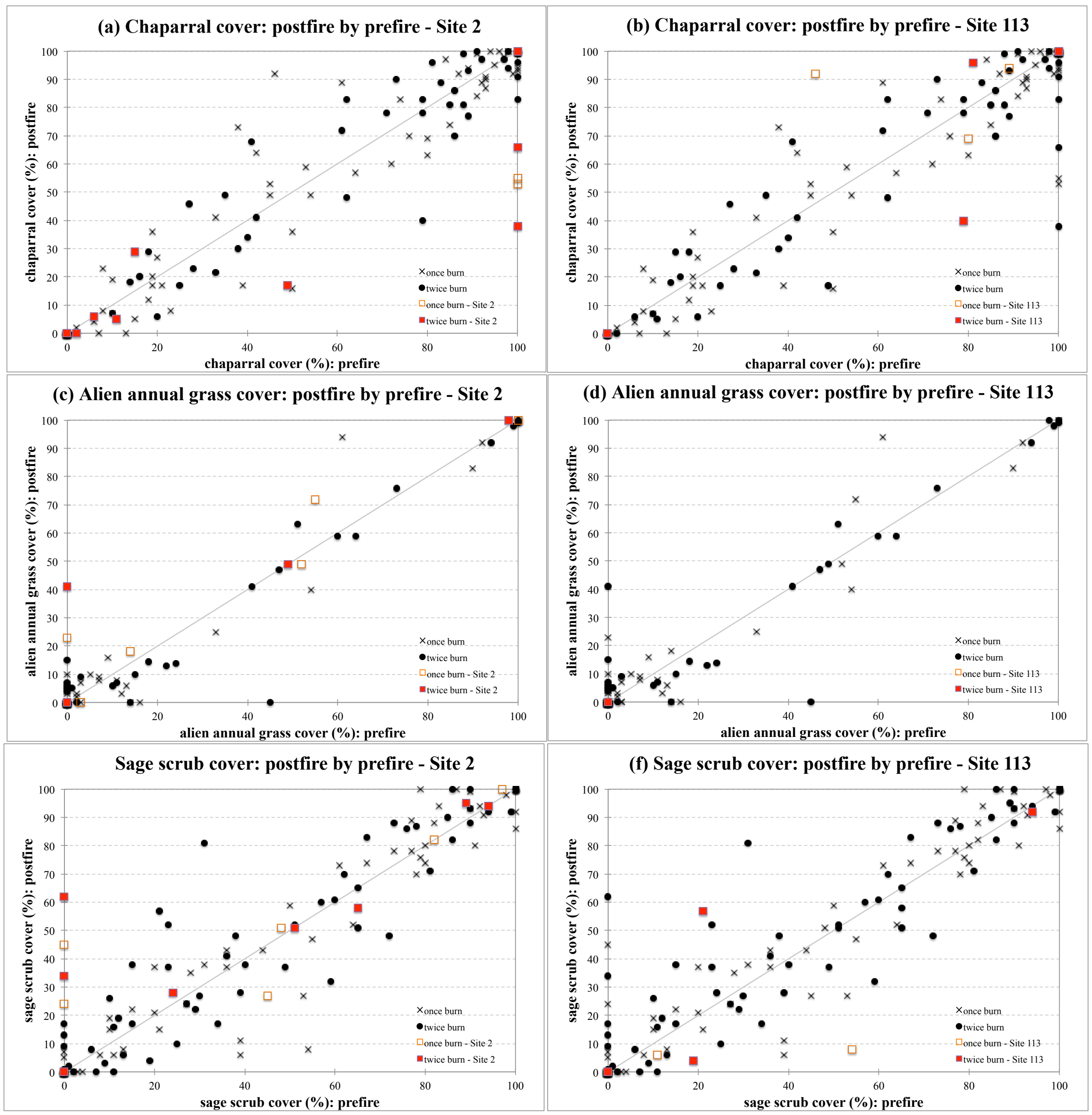



**Figure 11. Postfire by prefire vegetation cover with subsites from Site 2 (a, c, e) and Site 113 (b, d, f) highlighted. Gray line represents 1:1 ratio. a-b: chaparral cover, c-d: alien annual grass cover, e-f: sage scrub cover.**



# VIII. TABLES

**Table 1. Total area of minimum fire intervals in Ventura (5,720 $km^2$) and Los Angeles (12,310 $km^2$) 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 |

**Table 2. Details of the twelve sites with once burn and twice burn polygons in Ventura and Los Angeles Counties. Minimum Fire Interval from Cal FRAP (1898-2007). Distance from coast (km) calculated in ArcGIS 10.1 (NAD1983 UTM). * indicates sites located within the Santa Ana wind corridors.**

| | SITE ID | PREFIRE IMAGE | MINIMUM FIRE INTERVAL | POSTFIRE IMAGE | YEARS OF REGROWTH | DISTANCE FROM COAST (km) (once burn, twice burn) |
|---|---|---|---|---|---|---|
| 1 | 001 | June 30, 1961 | 1962<br>1967 | 1975 | 8 | 26.62, 25.54 |
| 2 | 002 | 1994 | 1998<br>2003 | 2009 | 6 | 44.22, 40.36 |
| 3 | 004 | June 30, 1961 | 1962<br>1967 | 1980 | 13 | 21.98, 17.17 |
| 4 | 006 | February 17, 1964 | July 1, 1985<br>June 26, 1990 | 2009 | 19 | 21.60, 20.42 |
| 5 | 102* | September 7, 1990 | November 2, 1993<br>October 21, 1996 | 2005 | 9 | 6.38, 2.54 |
| 6 | 103 | Aug. 23-24, 1956 | 1975<br>1979 | June 6, 2002 | 23 | 46.34, 44.67 |
| 7 | 104 | August 11, 1959 | 1960<br>1964 | June 13, 2002 | 38 | 61.12, 60.33 |
| 8 | 109 | August 12, 1959 | 1967<br>1970 | August 25, 1980 | 10 | 35.00, 36.00 |
| 9 | 110* | February 14, 1964 | 1967<br>1970 | March 14, 1977 | 7 | 20.79, 21.79 |
| 10 | 111* | March 12, 1976 | November 24,1980<br>October 9,1982 | June 1, 1994 | 12 | 8.36, 8.83 |
| 11 | 112 | August 24, 1978 | 1985<br>1990 | 2009 | 19 | 9.91, 11.94 |
| 12 | 113* | November 3, 1952 | 1956<br>1958 | October 18, 1980 | 22 | 6.48, 4.67 |

**Table 3. Rate of chaparral change (%) by time since wildfire for once burn (n = 12) or twice burn (n = 12) polygons on north and south aspects. Linear regressions and $R^2$ values calculated in Excel.**

| Aspect | Burn History | Calculated with 12 sites | Calculated with 11 sites (omitting Site 104) |
|---|---|---|---|
| north | once burn | $y = 0.4479x - 9.7592$<br>$R^2 = 0.28$ | $y = 0.4137x - 9.3481$<br>$R^2 = 0.12038$ |
| | twice burn | $y = 0.4977x - 11.484$<br>$R^2 = 0.22378$ | $y = 0.8331x - 15.516$<br>$R^2 = 0.2681$ |
| south | once burn | $y = 0.174x - 1.4639$<br>$R^2 = 0.1107$ | $y = 0.5008x - 5.3926$<br>$R^2 = 0.38008$ |
| | twice burn | $y = 0.0163x + 2.5937$<br>$R^2 = 0.00058$ | $y = 0.2201x + 0.1429$<br>$R^2 = 0.04428$ |
| north & south | once burn | $y = 0.3817x - 6.5503$<br>$R^2 = 0.51238$ | $y = 0.4742x - 7.6613$<br>$R^2 = 0.39711$ |
| | twice burn | $y = 0.1947x - 4.033$<br>$R^2 = 0.11785$ | $y = 0.2458x - 4.7779$<br>$R^2 = 0.08065$ |

# APPENDIX A: SITE DETAILS

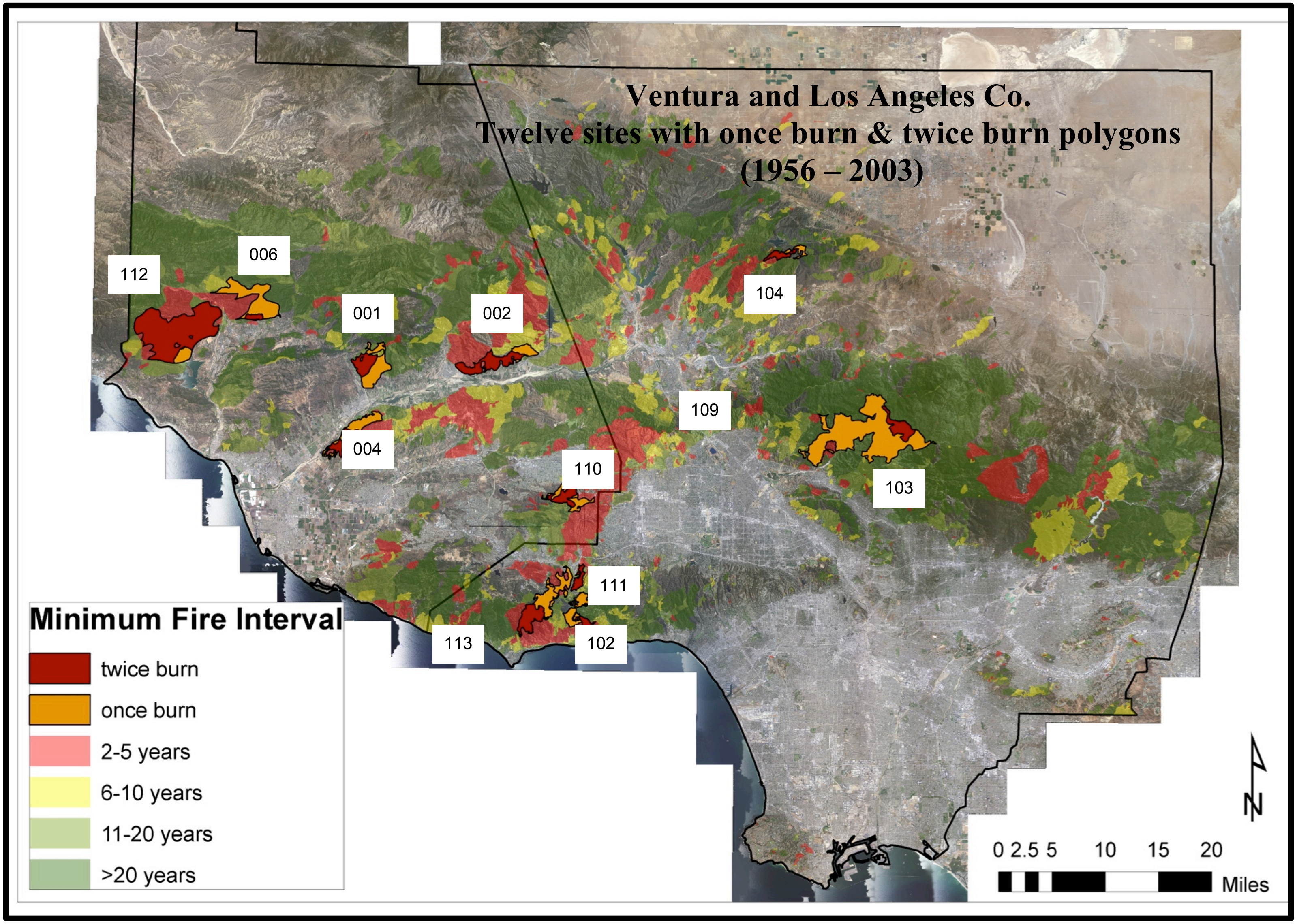
1. Number of subsites and equivalent areas ($m^2$) per polygon.

| | SITE ID | once burn polygon | | | twice burn polygon | | |
|---|---|---|---|---|---|---|---|
| | | Number of subsites | Total area analyzed ($m^2$) | Area equivalent to 1% ($m^2$) | Number of subsites | Total area analyzed ($m^2$) | Area equivalent to 1% ($m^2$) |
| 1 | 001 | 9 | 22,500 | 225 | 6 | 15,000 | 150 |
| 2 | 002 | 8 | 20,000 | 200 | 9 | 22,500 | 225 |
| 3 | 004 | 8 | 20,000 | 200 | 10 | 25,000 | 250 |
| 4 | 006 | 7 | 17,500 | 175 | 9 | 22,500 | 225 |
| 5 | 102 | 8 | 20,000 | 200 | 9 | 22,500 | 225 |
| 6 | 103 | 10 | 25,000 | 250 | 10 | 25,000 | 250 |
| 7 | 104 | 6 | 15,000 | 150 | 7 | 17,500 | 175 |
| 8 | 109 | 9 | 22,500 | 225 | 5 | 12,500 | 125 |
| 9 | 110 | 9 | 22,500 | 225 | 9 | 22,500 | 225 |
| 10 | 111 | 8 | 20,000 | 200 | 7 | 17,500 | 175 |
| 11 | 112 | 9 | 22,500 | 225 | 10 | 25,000 | 250 |
| 12 | 113 | 8 | 20,000 | 200 | 8 | 20,000 | 200 |

2. Wildfire details for twelve sites with once burn and twice burn polygons. Site location from Google Earth. Fire history from Cal FRAP (1898-2007). [] indicate wildfires included in this study. Polygon area calculated in ArcGIS 10.1 (NAD1983 UTM).

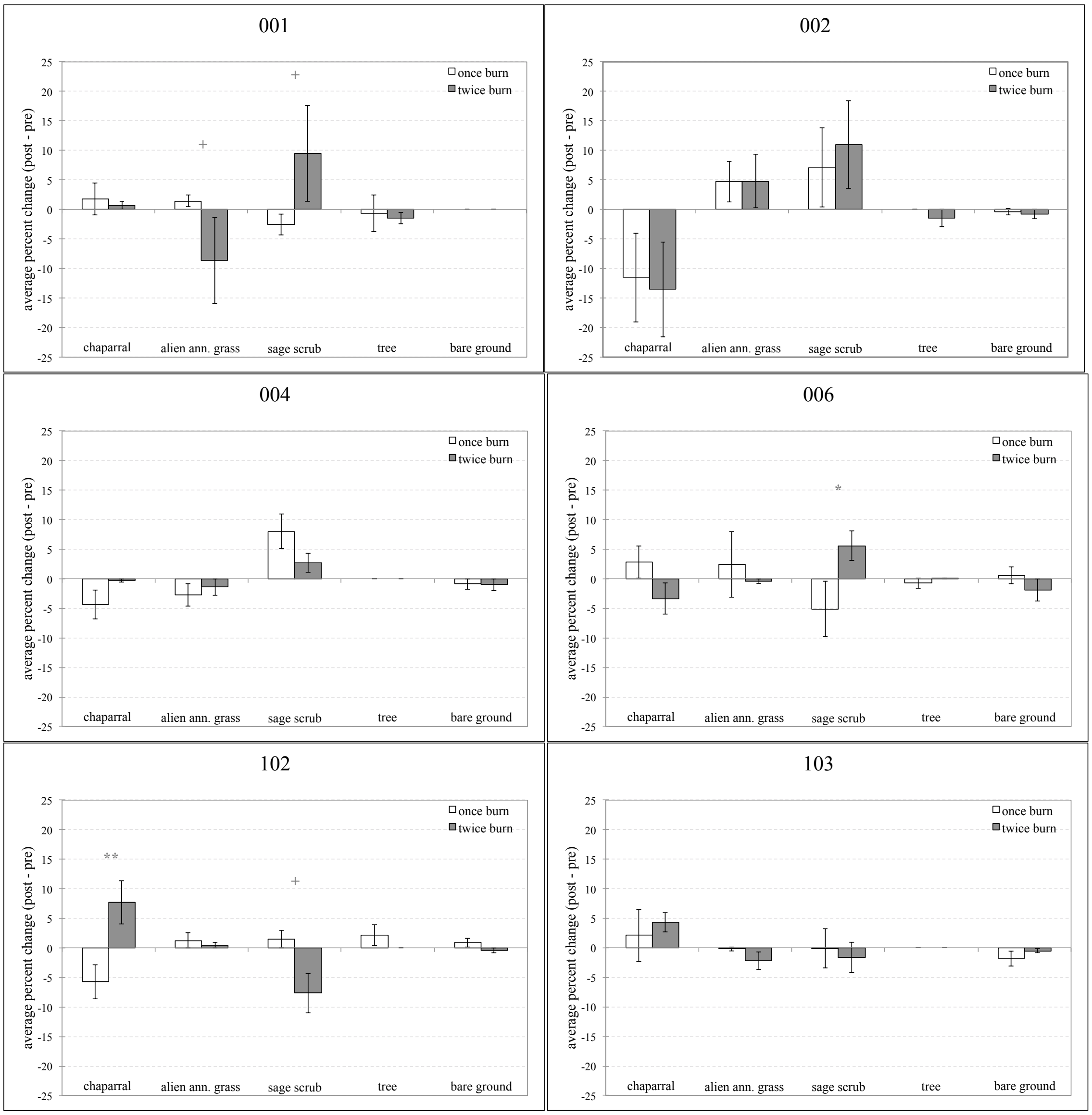
| | SITE ID | SITE LOCATION | SITE LOCATION | FIRE HISTORY | AREA: ONCE BURN ($km^2$) | AREA: TWICE BURN ($km^2$) |
|---|---|---|---|---|---|---|
| 1 | 001 | East of 150 | 34º24’N, 119º03’W | 1936, [1962, 1967], 1985 | 13.31 | 7.80 |
| 2 | 002 | Fillmore | 34º25’N, 118º49’W | 1918, 1922, 1954, 1983, [1998, 2003] | 4.76 | 14.61 |
| 3 | 004 | Santa Paula | 34º18’N, 119º05’W | [1962, 1967], 1986, 2003 | 10.18 | 5.48 |
| 4 | 006 | Ojai | 34º28’N, 119º14’W | 1917, 1948, [1985, 1990] | 23.05 | 1.99 |
| 5 | 102 | Santa Monica Mts.: Malibu Canyon Road | 34º04’N, 118º41’W | 1942, 1970, [1993, 1996], 2007 | 8.08 | 3.54 |
| 6 | 103 | East of San Fernando | 34º20’N, 118º11’W | 1956, [1975, 1979], 1998 | 77.63 | 8.50 |
| 7 | 104 | Bouquet Reservoir | 34º34’N, 118º22’W | 1923, 1937, [1960, 1964], 2007 | 2.35 | 3.47 |
| 8 | 109 | Santa Clarita | 34º21’N, 118º31’ W | 1913, [1967, 1970], 1983, 1989, 1997, 2007 | 1.90 | 0.53 |
| 9 | 110 | Simi Valley | 34º14’N, 118º44’ W | [1967, 1970], 1982 | 6.42 | 6.78 |
| 10 | 111 | Santa Monica Mts.: Lost Hills Road | 34º07’N, 118º42’ W | 1958, 1970, [1980, 1982], 1996 | 8.13 | 3.71 |
| 11 | 112 | Lake Casitas | 34º26’N, 119º21’W | 1898, 1910, 1917, 1923, 1932, [1985, 1990] | 66.67 | 4.39 |
| 12 | 113 | Santa Monica Mts.: Bulldog Motorway | 34º05’N, 118º46’W | 1935, [1956, 1958], 1982 | 8.68 | 10.66 |

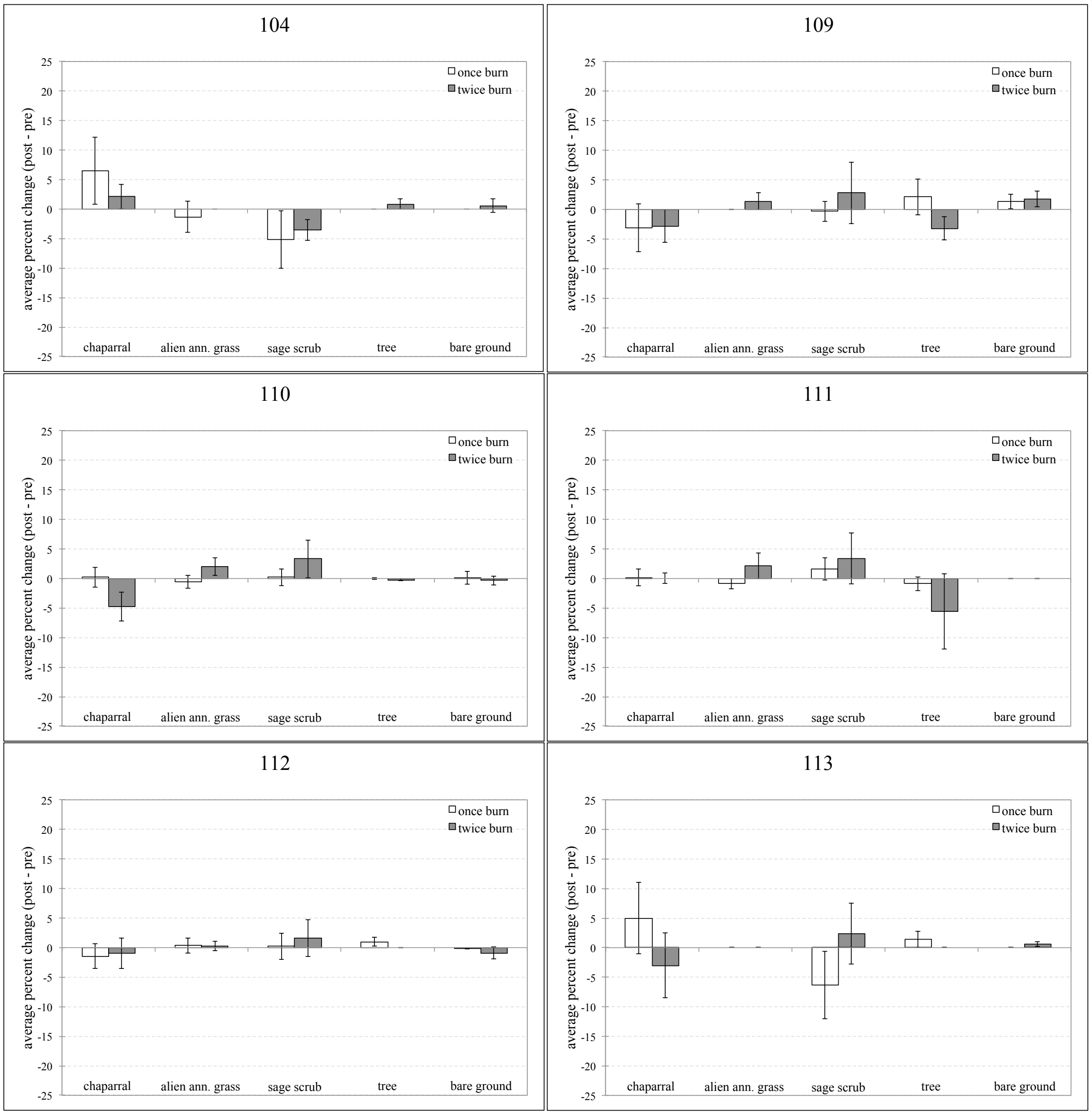
3. Geographic location of the twelve once burn and twice burn polygons in Ventura or Los Angeles County.



## APPENDIX B: AVERAGE VEGETAION CHANGE BY SITE

1. Average vegetation change (%) per site following one or two wildfires within the same five year period ± one standard error. A value of 0 indicates “no change” from prefire to postfire vegetation cover. Significance determined by applying a negative binomial regression analysis to subsite data: + p-value < 0.08, * p-value < 0.05, ** p-value < 0.01.

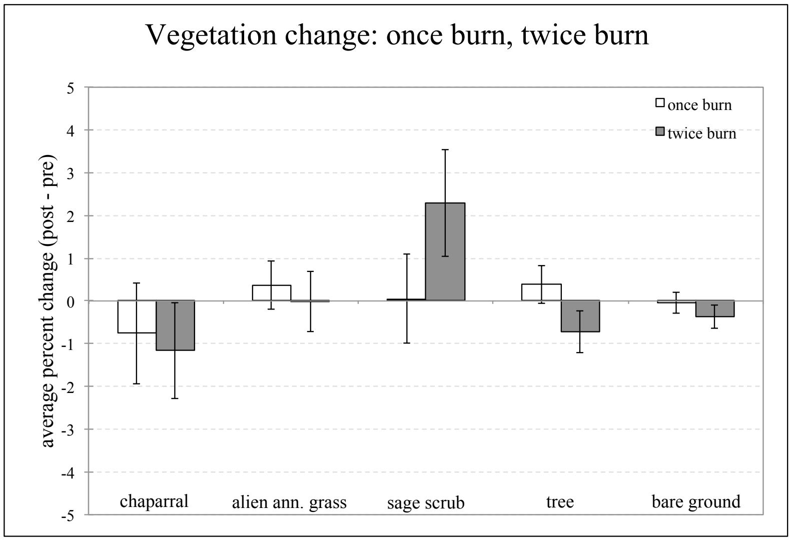





Inspecting each site individually, nine of the twelve sites showed no significant differences in vegetation change following a single short interval wildfire (twice burn) compared to a longer, more historic, wildfire interval (once burn). Site 1, located near Santa Paula in Ventura County, showed strong differences in alien annual grass cover (p = 0.076) and sage scrub cover (p = 0.052) between once and twice burned polygons: alien annual grass cover was lower and sage scrub cover was greater in the twice burn polygon. Site 6, located near Ojai, CA and also in Ventura County, showed a greater increase in sage scrub cover (p = 0.035) in the twice burn polygon compared to the once burned polygon. Site 102, located in the Santa Monica Mountains in Los Angeles County, was the only site to show a significant difference in delta chaparral cover (p = 0.006) between twice and once burned polygons with a greater increase in chaparral cover and a greater decline in sage scrub cover in the twice burn polygon. Each percent change in vegetation cover is approximately 206 $m^2$ (Appendix A).

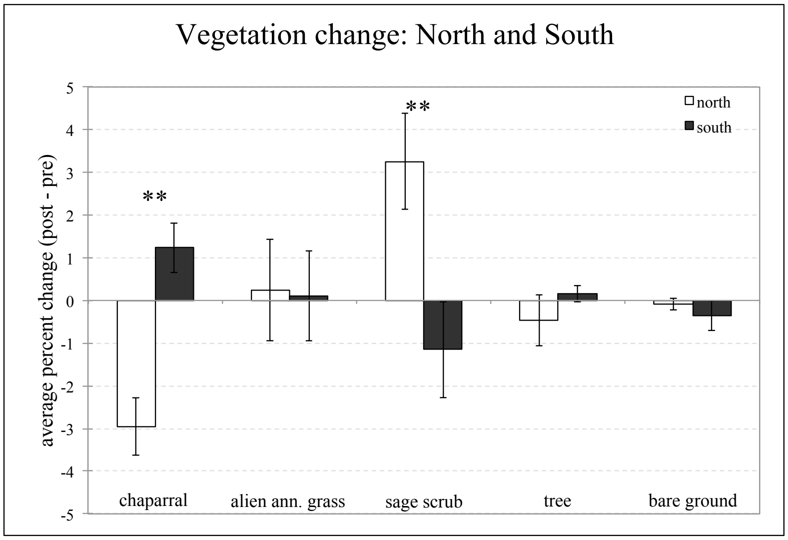
## APPENDIX C: SUPPLEMENTAL DATA

1. Average vegetation change (%) by burn history (once burn/twice burn) across 198 subsites ± standard error. Significance determined by a negative binomial regression analysis. Site was considered as a covariate in a negative binomial regression analysis and was not significant except for Site 2 (chaparral, alien annual grass). See Appendix B for site details.



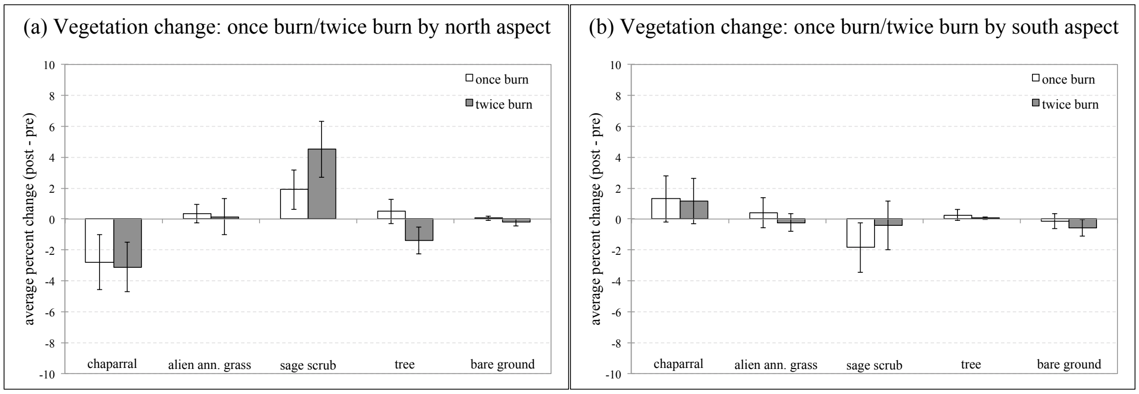

| Vegetation class | Burn history | Number of subsites | Mean (%) | Std. error | Min (%) | Median (%) | Max (%) | p-value |
|---|---|---|---|---|---|---|---|---|
| chaparral | once | 99 | -0.76 | 1.18 | -47 | 0 | 46 | 0.805 |
| | twice | 99 | -1.17 | 1.12 | -62 | 0 | 27 | |
| alien annual grass | once | 99 | 0.37 | 0.57 | -16 | 0 | 33 | 0.782 |
| | twice | 99 | -0.02 | 0.70 | -45 | 0 | 41 | |
| sage scrub | once | 99 | 0.05 | 1.04 | -46 | 0 | 45 | 0.163 |
| | twice | 99 | 2.28 | 1.24 | -27 | 0 | 62 | |
| tree | once | 99 | 0.38 | 0.44 | -22 | 0 | 24 | 0.434 |
| | twice | 99 | -0.73 | 0.48 | -44 | 0 | 6 | |
| bare grd/ exposed rock | once | 99 | -0.05 | 0.25 | -9 | 0 | 11 | 0.82 |
| | twice | 99 | -0.37 | 0.27 | -14 | 0 | 7 | |

2. Average vegetation change (%) by aspect (north/south) across 198 subsites ± standard error. Significance determined by a negative binomial regression analysis. Site was considered as a covariate and was not significant except for Site 2 (chaparral, alien annual grass) and Site 104 (sage scrub). See Appendix B for site details. ** p-value < 0.01



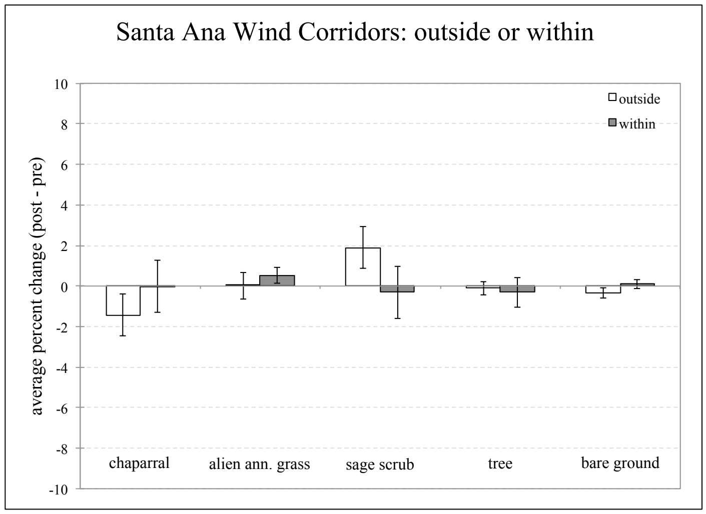

| Vegetation class | Aspect | Number of subsites | Mean (%) | Std. error | Min (%) | Median (%) | Max (%) | p-value |
|---|---|---|---|---|---|---|---|---|
| chaparral | north | 104 | -2.95 | 0.68 | -62 | 0 | 28 | 0.00962 |
| | south | 94 | 1.23 | 0.058 | -39 | 0 | 46 | |
| alien annual grass | north | 104 | 0.24 | 1.19 | -45 | 0 | 41 | 0.925 |
| | south | 94 | 0.11 | 1.05 | -16 | 0 | 33 | |
| sage scrub | north | 104 | 3.26 | 1.13 | -28 | 0 | 62 | 0.00529 |
| | south | 94 | -1.15 | 1.13 | -46 | 0 | 36 | |
| tree | north | 104 | -0.47 | 0.60 | -44 | 0 | 24 | 0.657 |
| | south | 94 | 0.16 | 0.20 | -7 | 0 | 14 | |
| bare grd/ exposed rock | north | 104 | -0.09 | 0.14 | -10 | 0 | 7 | 0.852 |
| | south | 94 | -0.35 | 0.35 | -14 | 0 | 11 | |

3. Average vegetation change (%) by fire history on (a) north or (b) south aspect ± one standard error across 198 subsites. Significance determined by a negative binomial regression analysis. Site was considered as a covariate and was not significantly different except for Site 2 (chaparral, alien annual grass, sage scrub), Site 104 (sage scrub), and Site 109 (chaparral). See Appendix B for site details.




| Aspect | Vegetation Class | once burn polygon (n = 50) | | twice burn polygon (n = 54) | | |
|---|---|---|---|---|---|---|
| | | Mean | Std. error | Mean | Std. error | p-value |
| north | chaparral | -2.78 | 1.78 | -3.10 | 1.60 | 0.6625 |
| | alien annual grass | 0.34 | 0.60 | 0.16 | 1.19 | 0.9678 |
| | sage scrub | 1.9 | 1.27 | 4.52 | 1.82 | 0.1209 |
| | tree | 0.5 | 0.79 | -1.37 | 0.88 | 0.325 |
| | bare grd/ exposed rock | 0.04 | 0.15 | -0.20 | 0.24 | 0.900 |
| | | once burn polygon (n = 49) | | twice burn polygon (n = 45) | | |
| south | chaparral | 1.31 | 1.50 | 1.16 | 1.48 | 0.919 |
| | alien annual grass | 0.41 | 0.97 | -0.22 | 0.60 | 0.759 |
| | sage scrub | -1.84 | 1.61 | -0.40 | 1.58 | 0.566 |
| | tree | 0.27 | 0.37 | 0.04 | 0.07 | 0.949 |
| | bare grd/ exposed roc | -0.14 | 0.47 | -0.58 | 0.53 | 0.901 |

4. Average vegetation change (%) by location in relation to the Santa Ana wind corridors (outside/within) across 198 subsites. Site was considered as a covariate and was not significantly different expect for Site 2 (chaparral, alien annual grass) outside the Santa Ana wind corridors. See Appendix B for site details. Significance determined by a negative binomial regression analysis. Tables below include (a) mean values ± standard error calculated across 198 subsites, (b) mean values ± standard error calculated across 24 polygons, (c) detailed subsite results (n = 198), and (d) detailed polygon (n = 24) results.




(a) Subsite

| Santa Ana wind corridors | Average percent change (post – pre) (n = 198 subsites) | | | | |
|---|---|---|---|---|---|
| | chaparral | alien annual grass | sage scrub | tree | bare grd/ exposed rock |
| outside n = 132 | -1.43 ± 1.03 | 0.00 ± 0.65 | 1.90 ± 1.03 | -0.11 ± 0.33 | -0.36 ± 0.25 |
| within n = 66 | -0.02 ± 1.30 | 0.53 ± 0.39 | -0.30 ± 1.30 | -0.30 ± 0.74 | 0.09 ± 0.21 |

(b) Polygon

| Santa Ana wind corridors | Average percent change (post – pre) (n = 24 polygons) | | | | |
|---|---|---|---|---|---|
| | chaparral | alien annual grass | sage scrub | tree | bare grd/ exposed rock |
| outside n = 16 | -1.32 ± 1.32 | -0.09 ± 0.78 | 1.88 ± 1.28 | -0.22 ± 0.30 | -0.25 ± 0.25 |
| within n = 8 | -0.06 ± 1.63 | 0.55 ± 0.40 | -0.19 ± 1.54 | -0.40 ± 0.81 | 0.10 ± 0.16 |

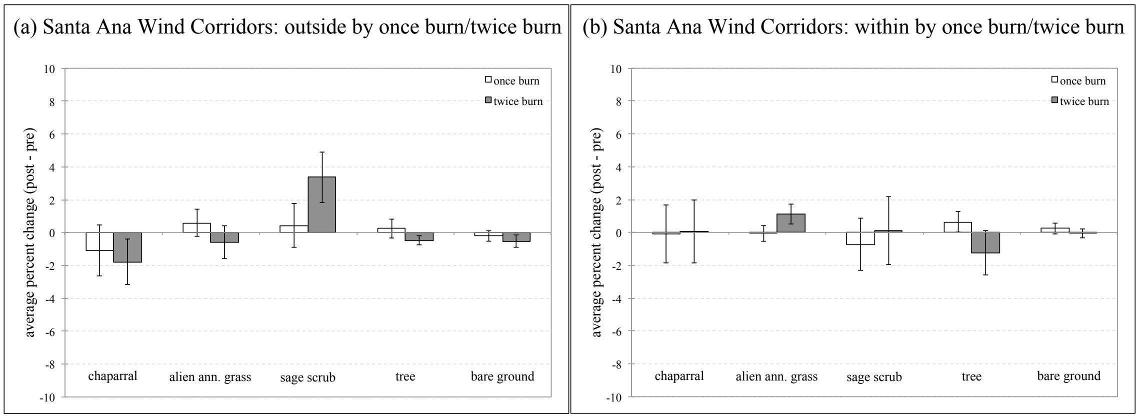
(c) Subsite

| Vegetation class | Santa Ana | Number of subsites | Mean (%) | Std. error | Min (%) | Median (%) | Max (%) | p-value |
|---|---|---|---|---|---|---|---|---|
| chaparral | outside | 132 | -1.43 | 1.03 | -62 | 0 | 35 | 0.415 |
| | within | 66 | -0.02 | 1.30 | -39 | 0 | 46 | |
| alien annual grass | outside | 132 | 0.00 | 0.65 | -45 | 0 | 41 | 0.725 |
| | within | 66 | 0.53 | 0.39 | -9 | 0 | 15 | |
| sage scrub | outside | 132 | 1.90 | 1.03 | -33 | 0 | 62 | 0.194 |
| | within | 66 | -0.30 | 1.30 | -46 | 0 | 36 | |
| tree | outside | 132 | -0.11 | 0.33 | -22 | 0 | 24 | 0.896 |
| | within | 66 | -0.30 | 0.74 | -44 | 0 | 14 | |
| bare grd/ exposed rock | outside | 132 | -0.36 | 0.25 | -14 | 0 | 11 | 0.763 |
| | within | 66 | 0.09 | 0.21 | -6 | 0 | 7 | |

(d) Polygon

| Vegetation class | Santa Ana | Number of polygons | Mean (%) | Std. error | Min (%) | Median (%) | Max (%) | p-value |
|---|---|---|---|---|---|---|---|---|
| chaparral | outside | 16 | -1.32 | 1.32 | -13.56 | 0 | 6.50 | 0.9317 |
| | within | 8 | -0.06 | 1.63 | -5.75 | 0 | 7.67 | |
| alien annual grass | outside | 16 | -0.09 | 0.78 | -8.67 | 0 | 4.78 | 0.4666 |
| | within | 8 | 0.55 | 0.40 | -0.88 | 0 | 2.14 | |
| sage scrub | outside | 16 | 1.88 | 1.28 | -5.17 | 0 | 11.00 | 0.2685 |
| | within | 8 | -0.19 | 1.54 | -7.67 | 0 | 3.43 | |
| tree | outside | 16 | -0.22 | 0.30 | -3.20 | 0 | 2.11 | 0.638 |
| | within | 8 | -0.40 | 0.81 | -5.57 | 0 | 2.13 | |
| bare grd/ exposed rock | outside | 16 | -0.25 | 0.25 | -1.89 | 0 | 1.80 | 0.931 |
| | within | 8 | 0.10 | 0.16 | -0.44 | 0 | 0.88 | |

5. Average vegetation change (%) by burn history (once burn/twice burn) (a) outside or (b) within the Santa Ana wind corridors across 198 subsites. Site was considered as a covariate and was not significantly different expect for Site 2 (chap., alien annual grass) outside the Santa Ana wind corridors. See Appendix B for site details. Significance determined by negative binomial regression analysis.

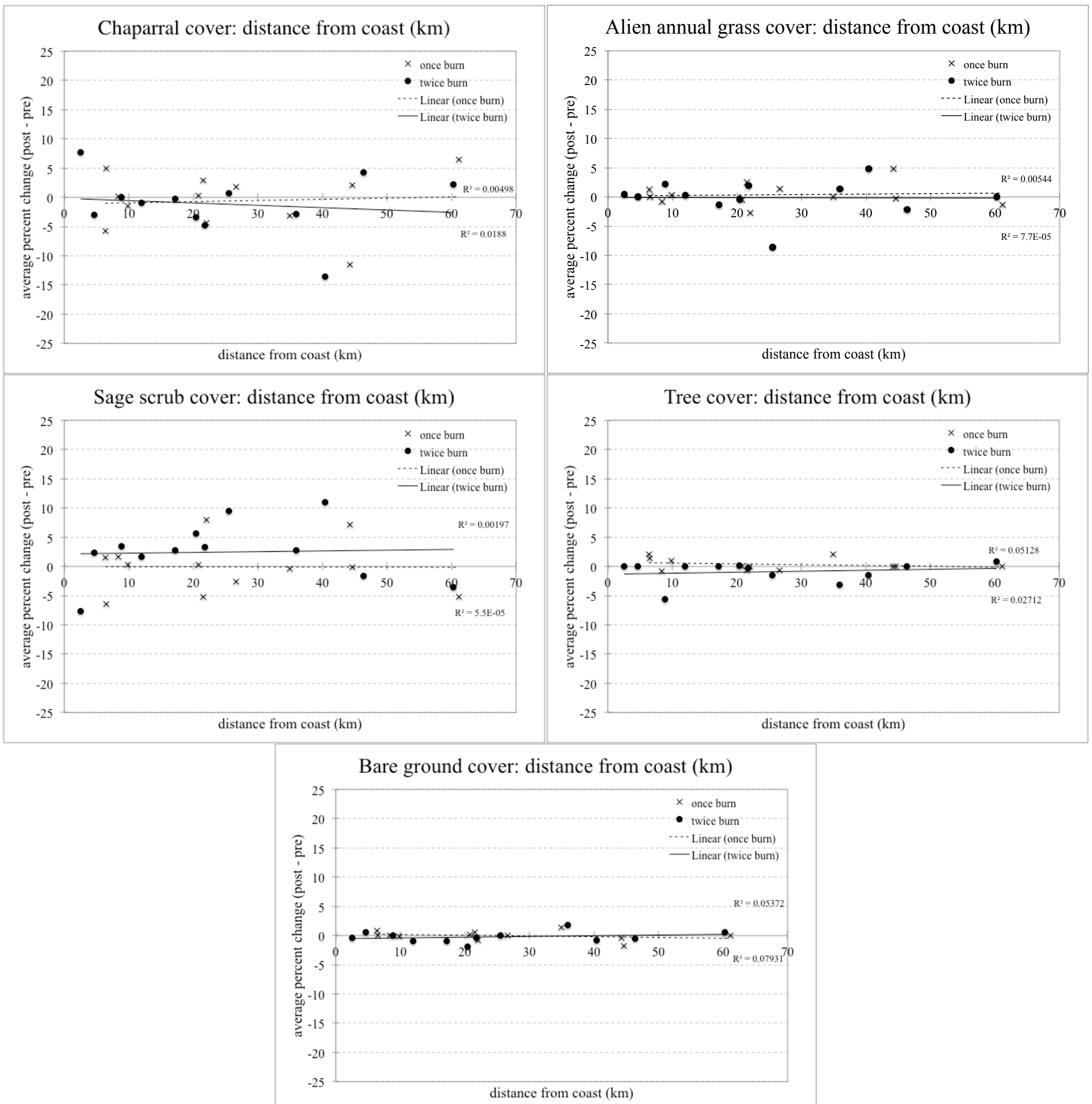



| Santa Ana wind corridors | Vegetation Class | once burn subsites (n = 66) | | twice burn subsites (n = 66) | | |
|---|---|---|---|---|---|---|
| | | Mean (%) | Std. error | Mean (%) | Std. error | p-value |
| outside | chaparral | -1.091 | 1.541 | -1.778 | 1.375 | 0.746 |
| | alien annual grass | 0.591 | 0.816 | -0.584 | 1.000 | 0.497 |
| | sage scrub | 0.439 | 1.344 | 3.365 | 1.544 | 0.147 |
| | tree | 0.258 | 0.585 | -0.473 | 0.290 | 0.676 |
| | bare ground/ exposed rock | -0.197 | 0.326 | -0.530 | 0.386 | 0.848 |
| | | once burn subsites (n = 33) | | twice burn subsites (n = 33) | | |
| within | chaparral | -0.091 | 1.769 | 0.061 | 1.924 | 0.953 |
| | alien annual grass | -0.061 | 0.477 | 1.121 | 0.610 | 0.632 |
| | sage scrub | -0.727 | 1.580 | 0.121 | 2.081 | 0.742 |
| | tree | 0.636 | 0.615 | -1.242 | 1.341 | 0.445 |
| | bare ground/ exposed rock | 0.242 | 0.340 | -0.061 | 0.265 | 0.902 |

6. Average vegetation change (%) by aspect (north/south) outside or within the Santa Ana wind corridors across 198 subsites. Site was considered as a covariate and was not significantly different expect for Site 2 (chaparral, alien annual grass) and Site 104 (sage scrub) outside the Santa Ana wind corridors. See Appendix B for site details. Significant determined by a negative binomial regression analysis: * p-value < 0.05, + p-value < 0.1.

| **Santa Ana wind corridors** | **Vegetation Class** | **north (n = 70)** | | **south (n = 62)** | | |
|---|---|---|---|---|---|---|
| | | **Mean (%)** | **Std. error** | **Mean (%)** | **Std. error** | **p-value** |
| outside | chaparral | -3.391 | 1.541 | 0.774 | 1.375 | 0.045* |
| | alien annual grass | 0.092 | 0.816 | -0.097 | 1.000 | 0.917 |
| | sage scrub | 3.802 | 1.344 | -0.242 | 1.544 | 0.043* |
| | tree | -0.360 | 0.585 | 0.177 | 0.290 | 0.759 |
| | bare ground/ exposed rock | -0.143 | 0.326 | -0.613 | 0.386 | 0.787 |
| | | **north (n = 34)** | | **south (n = 32)** | | |
| within | chaparral | -2.029 | 0.802 | 2.125 | 2.502 | 0.100+ |
| | alien annual grass | 0.559 | 0.453 | 0.500 | 0.655 | 0.981 |
| | sage scrub | 2.147 | 1.073 | -2.906 | 2.356 | 0.043* |
| | tree | -0.706 | 1.442 | 0.125 | 0.117 | 0.735 |
| | bare ground/ exposed rock | 0.029 | 0.029 | 0.156 | 0.445 | 0.959 |

7. Average vegetation change (%) by distance from coast across 12 sites (12 once burn polygons, 12 twice burn polygons). Significance determined by linear regression analysis in JMP. Distances (km) are in Table 2. Tables (a) and (b) include average vegetation change by fire history or by overall trends regardless of fire history. Tables (c) and (d) include average vegetation change standardized by the number of postfire years by fire history or by overall trends regardless of fire history.



a) Average vegetation change by fire history

| Vegetation Class | once burn polygons (n = 12) | | | twice burn polygons (n = 12) | | |
|---|---|---|---|---|---|---|
| | $R^2$ | coefficient | p-value | $R^2$ | coefficient | p-value |
| chaparral | 0.0050 | 0.0201 | 0.8275 | 0.0188 | -0.0403 | 0.6709 |
| alien annual grass | 0.0054 | 0.0081 | 0.8197 | 0.0001 | -0.0016 | 0.9784 |
| sage scrub | 0.0001 | -0.0018 | 0.9818 | 0.0020 | 0.0128 | 0.8909 |
| tree | 0.0513 | -0.0135 | 0.4791 | 0.02712 | 0.0168 | 0.6090 |
| bare ground/ exposed rock | 0.0793 | -0.0128 | 0.3752 | 0.0537 | 0.0123 | 0.4685 |

b) Average vegetation change by overall trends regardless of fire history

| Vegetation Class | Overall trend: once burn & twice burn polygons (n = 24) | | |
|---|---|---|---|
| | $R^2$ | coefficient | p-value |
| chaparral | 0.0012 | -0.0101 | 0.8711 |
| alien annual grass | 0.0006 | 0.0036 | 0.9124 |
| sage scrub | 0.0002 | 0.0035 | 0.9540 |
| tree | 0.0010 | 0.0029 | 0.8823 |
| bare ground/exposed rock | <0.0001 | 0.0001 | 0.9926 |

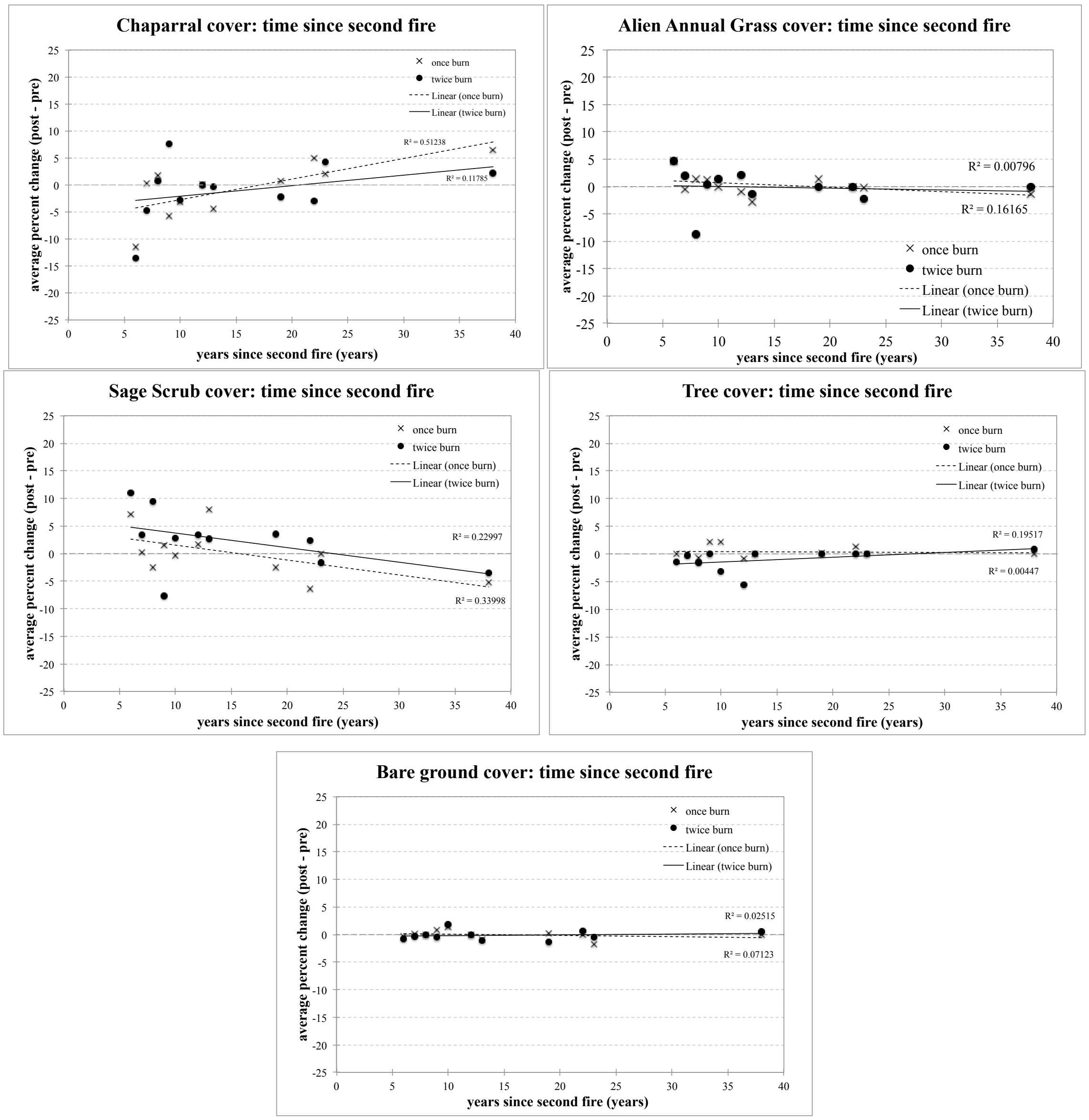
c) Standardized average vegetation change by fire history

| Vegetation Class | Standardized once burn polygons (n = 12) | | | Standardized twice burn polygons (n = 12) | | |
|---|---|---|---|---|---|---|
| | $R^2$ | coefficient | p-value | $R^2$ | coefficient | p-value |
| chaparral | 0.0331 | -0.006 | 0.57 | 0.0970 | -0.013 | 0.32 |
| alien annual grass | 0.0661 | 0.004 | 0.42 | 0.0072 | 0.002 | 0.79 |
| sage scrub | 0.0333 | 0.004 | 0.57 | 0.0720 | 0.010 | 0.40 |
| tree | 0.0285 | -0.001 | 0.60 | 0.0004 | <0.000 | 0.95 |
| bare ground/ exposed rock | 0.0678 | -0.001 | 0.41 | 0.0227 | 0.001 | 0.64 |

d) Standardized average vegetation change by overall trends regardless of fire history

| Vegetation Class | Standardized overall trend: once burn & twice burn polygons (n = 24) | | |
|---|---|---|---|
| | $R^2$ | coefficient | p-value |
| chaparral | 0.0637 | -0.0096 | 0.2340 |
| alien annual grass | 0.0213 | 0.0029 | 0.4967 |
| sage scrub | 0.0483 | 0.0071 | 0.3023 |
| tree | 0.0010 | -0.0003 | 0.8838 |
| bare ground/ exposed rock | 0.0005 | -0.0001 | 0.9142 |

8. Average vegetation change (%) by time since fire across 12 sites (12 once burn polygons, 12 twice burn polygons). “Time since fire” is equivalent to Years of Regrowth in Table 2. Table (a) vegetation change by once burn and twice burn polygons. Table (b) overall vegetation change with combined once burn and twice burn polygon. Significance determined by linear regression analysis in JMP: * p-value < 0.05, + p-value < 0.06.



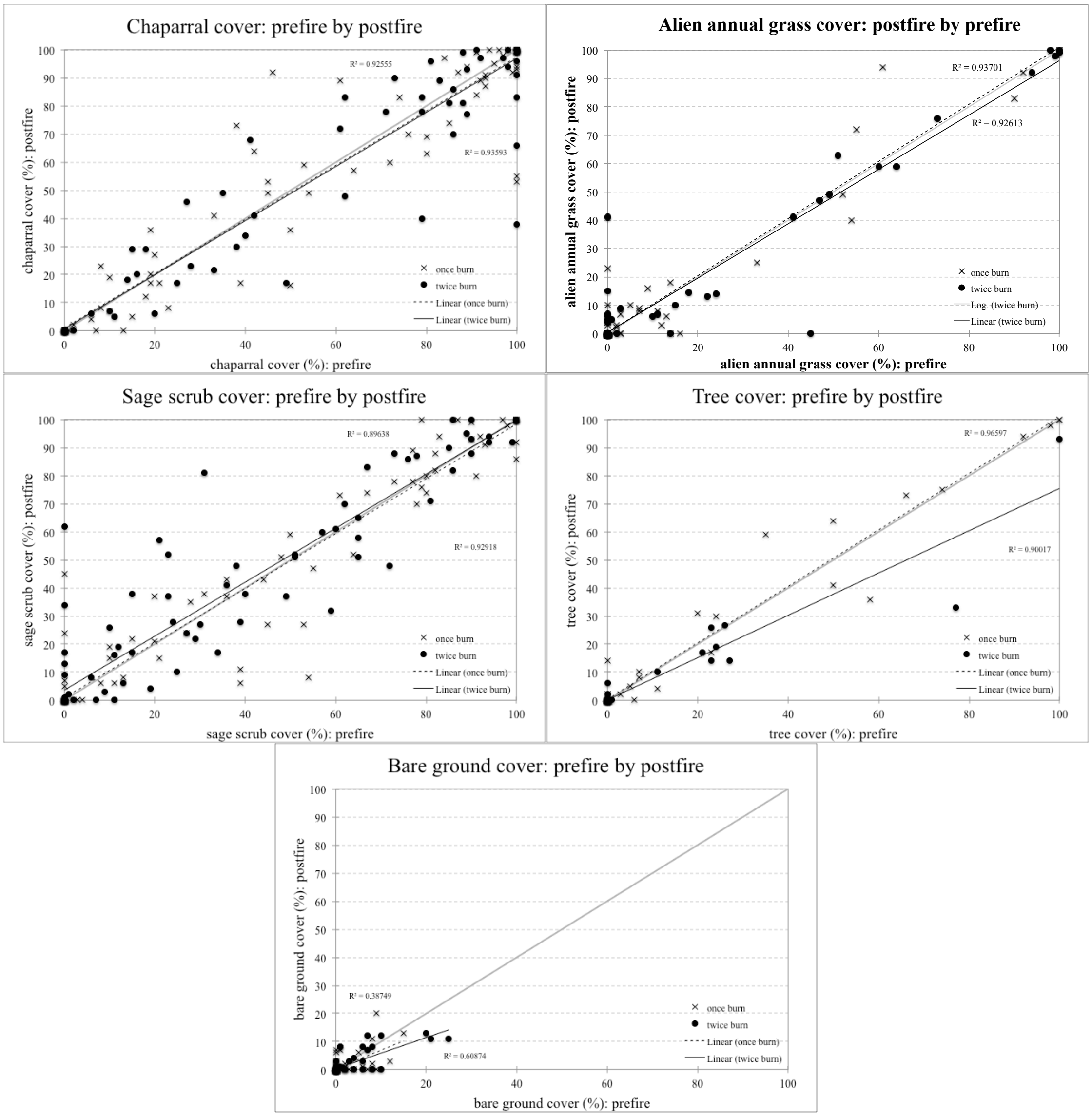
a) vegetation change by once burn and twice burn polygons

| Vegetation Class | once burn polygons (n = 12) | | | twice burn polygons (n = 12) | | |
|---|---|---|---|---|---|---|
| | $R^2$ | coefficient | p-value | $R^2$ | coefficient | p-value |
| chaparral | 0.50 | 0.38 | 0.01* | 0.11 | 0.19 | 0.29 |
| alien annual grass | 0.13 | -0.08 | 0.25 | 0.01 | -0.03 | 0.79 |
| sage scrub | 0.33 | -0.28 | 0.05+ | 0.21 | -0.26 | 0.13 |
| tree | <0.01 | -0.01 | 0.83 | 0.20 | 0.09 | 0.14 |
| bare ground/ exposed rock | 0.06 | -0.02 | 0.44 | 0.02 | 0.01 | 0.77 |

b) overall vegetation change with combined once burn and twice burn polygon

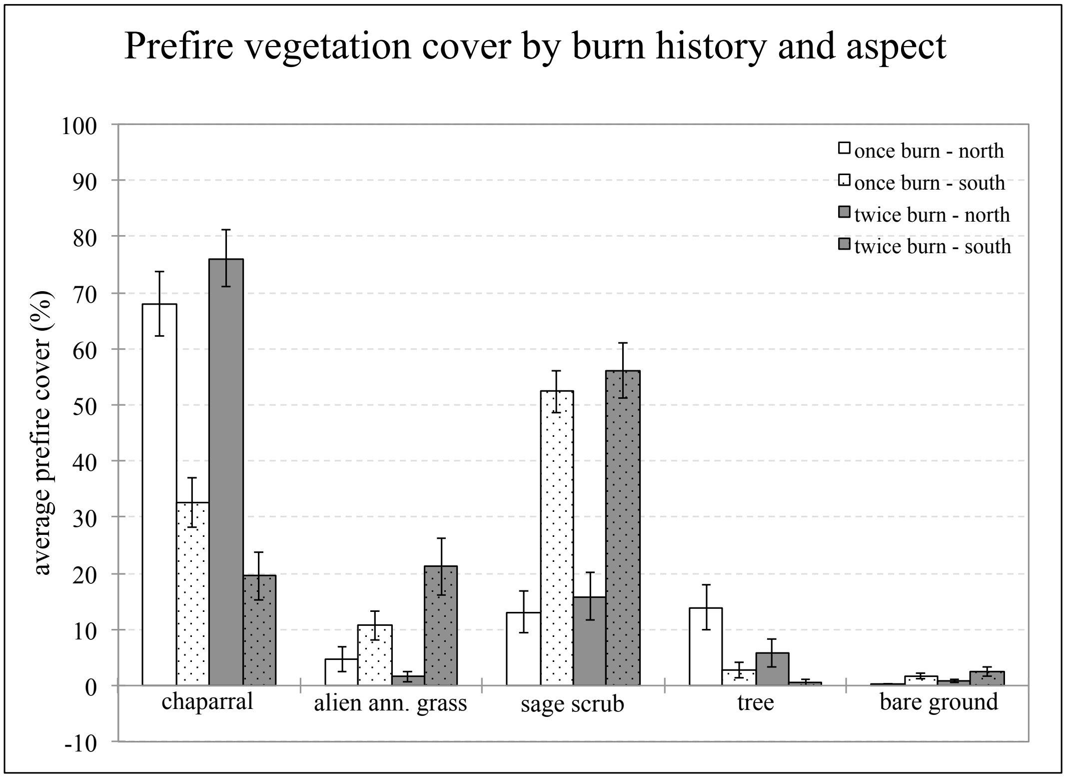
| Vegetation Type | Overall trend: once burn & twice burn polygons (n = 24) | | |
|---|---|---|---|
| | $R^2$ | coefficient | p-value |
| chaparral | 0.2635 | 0.2848 | 0.0103 |
| alien annual grass | 0.0337 | -0.0533 | 0.3903 |
| sage scrub | 0.2424 | -0.2662 | 0.0145 |
| tree | 0.0531 | 0.0405 | 0.2788 |
| bare ground/ exposed rock | 0.0038 | -0.0058 | 0.7762 |

9. Postfire by prefire vegetation cover (%) across 198 subsites. Significance determined by linear regression analysis in JMP.




| Vegetation Class | once burn subsites (n = 99) | | | twice burn subsites (n = 99) | | |
|---|---|---|---|---|---|---|
| | $R^2$ | coefficient | p-value | $R^2$ | coefficient | p-value |
| chaparral | 0.926 | 0.965 | <0.01 | 0.936 | 0.964 | <0.01 |
| alien annual grass | 0.937 | 1.007 | <0.01 | 0.926 | 0.959 | <0.01 |
| sage scrub | 0.929 | 0.976 | <0.01 | 0.896 | 0.962 | <0.01 |
| tree | 0.966 | 1.007 | <0.01 | 0.900 | 0.754 | <0.01 |
| bare ground/ exposed rock | 0.387 | 0.665 | <0.01 | 0.609 | 0.555 | <0.01 |

10. Prefire vegetation cover (%) by burn history (once burn/twice burn polygons) and aspect (north/south) for 198 subsites. Significance determined by a negative binomial regression analysis. Table below includes mean values ± one standard error.




| Burn History | Aspect | chaparral (% prefire) | alien annual grass (% prefire) | sage scrub (% prefire) | tree (% prefire) | bare grd./ exp. rock (% prefire) |
|---|---|---|---|---|---|---|
| once burn | north | 67.98 ± 5.70 | 4.70 ± 2.33 | 13.12 ± 3.70 | 13.92 ± 4.06 | 0.28 ± 0.13 |
| | south | 32.59 ± 5.33 | 10.6 ± 3.65 | 52.37 ± 5.48 | 2.71 ± 1.94 | 1.65 ± 0.52 |
| twice burn | north | 76.09 ± 5.01 | 1.59 ± 0.92 | 15.87 ± 4.18 | 5.72 ± 2.45 | 0.72 ± 0.27 |
| | south | 19.51 ± 4.26 | 21.22 ± 5.09 | 56.18 ± 4.96 | 0.53 ± 0.51 | 2.56 ± 0.88 |