

RESEARCH ARTICLE

## RESPONSES IN BIRD COMMUNITIES TO WILDLAND FIRES IN SOUTHERN CALIFORNIA

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### ABSTRACT

There is a growing body of literature covering the responses of bird species to wildland fire events. Our study was unique among these because we investigated the effects of large-scale wildland fires on entire bird communities across multiple vegetation types. We conducted avian point counts during the breeding seasons for two years before and two years after the Cedar and Otay Fires in 2003 in southern California. Our balanced sampling effort took place at two sites, one low-elevation and one high-elevation, each containing replicate stations (burned and unburned) within five vegetation types: chaparral, coastal sage scrub, grassland, oak woodland, and riparian. Although fire caused some degree of change in the vegetation structure at all of our impacted survey points, we found that the post-fire shrub and tree cover was significantly lower in only two of the vegetation types within the low-elevation site, coastal sage scrub and chaparral. We found no significant changes in cover at the high-elevation site. Using univariate and multivariate analyses, we tested whether the fires were associated with a change in bird species diversity, community structure, and the relative abundance of individuals within a species. We found that species diversity changed in only one circumstance: it increased in coastal sage scrub at the low-elevation site. Multivariate analyses revealed significant differences in the post-fire bird community structure in the low-elevation chaparral, low-elevation coastal sage scrub, and the high-elevation grassland communities. Vegetation characteristics altered by fire, such as decreases in shrub and tree cover, influenced the changes we observed in the bird communities. The relative abundance of some species (lazuli bunting [*Passerina amoena*] and horned lark [*Eremophila alpestris*]) significantly increased after the fires, while other species declined significantly (Anna's hummingbird [*Calypte anna*], wrentit [*Chamaea fasciata*], and bushtit [*Psaltriparus minimus*]). We detected mixed results for the spotted towhee (*Pipilo maculatus*), which increased in burned chaparral and declined in burned coastal sage scrub within the low-elevation site. We suggest that the observed responses of birds to these fires may be attributed to: (1) the availability of nearby unburned refugia, (2) the continued suitability of post-fire vegetation at the study sites, and (3) the generally high mobility of this taxon.

**Keywords:** bird communities, chaparral, coastal sage scrub, grassland, oak woodland, riparian, San Diego County, southern California, wildland fire

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## INTRODUCTION

Wildland fires have long been a part of the natural and human-altered environments in southern California, a fact that is unlikely to change anytime in the near future. Keeley *et al.* (1999, 2004) stated that large landscape-level fires occurred in the past and will continue as long as southern California experiences episodes of severe fire weather (e.g., fast, dry winds). Despite major fire suppression efforts and management plans, large and small fires remain frequent, either from natural ignition sources or those associated with the increasing human population (Keeley *et al.* 2004).

Fire can have both negative and positive impacts on the local flora and fauna. The native vegetation communities evolved with the regional fire regime and have adapted various survival and reproductive strategies in response to fire (Hanes 1971; Vogl and Schorr 1972; Keeley and Keeley 1981, 1984; Zedler *et al.* 1983; Keeley and Fotheringham 2001). However, as fire-return intervals decrease to below historic levels, the trend is for shrublands, whether chaparral or coastal sage scrub, to be type-converted to grasslands (Zedler *et al.* 1983, Keeley 2005). Just as fires alter the composition and structure of vegetation communities, animals may experience similar shifts in community structure and species occurrence. With the type-conversion of vegetation communities, we may expect a concomitant shift and potential loss of biodiversity in faunal populations. Fire may cause direct mortality or loss of habitat and food resources that result in the decline of some species (Whelan 1995). Other species that survive the fire and prefer open landscapes may benefit, and thereby increase in numbers.

Ecologists have described diverse and sometimes contrasting effects of fire on the avifauna of different vegetation communities in California (Lawrence 1966; Wirtz 1979, 1982; McClure 1981; Moriarty *et al.* 1985; Stanton 1986; Vreeland and Tietje 2002) and other regions of the United States (Fitzgerald and Tanner 1992, Kirkpatrick *et al.* 2006, Coppedge *et al.* 2008, Haney *et al.* 2008). As with other taxa, some birds may increase or decrease in population following fire depending largely on the life history of each species. In addition, the abundance of individuals within each species may change across successive post-fire years (Smith 2000, Marx *et al.* 2008).

In October and November of 2003, large landscape-level fires swept across southern California, consuming over 317 000 ha of wildlands. This included almost 130 000 ha in San Diego County that burned in the Cedar and Otay Fires. In addition to 15 human fatalities and the loss of nearly 5000 structures (CDF 2005), these two fires impacted the local plant and animal communities in a region already recognized as being at great risk for a loss of biodiversity (Mittermeier *et al.* 1997). Of particular concern was the fact that half of San Diego County's large network of protected habitat reserves, areas included in the Multiple Species Conservation Plan, was directly affected by the fires. Some of these reserves were entirely within the fire boundaries. Concern over the recovery of these wildlands and the species within them motivated our efforts to conduct this research.

We evaluated the response of breeding bird communities to fire by comparing data on species diversity, abundance, and community structure collected during the second and third years after the fires to existing data collected during two years of research before the fires. We examined bird response before and after

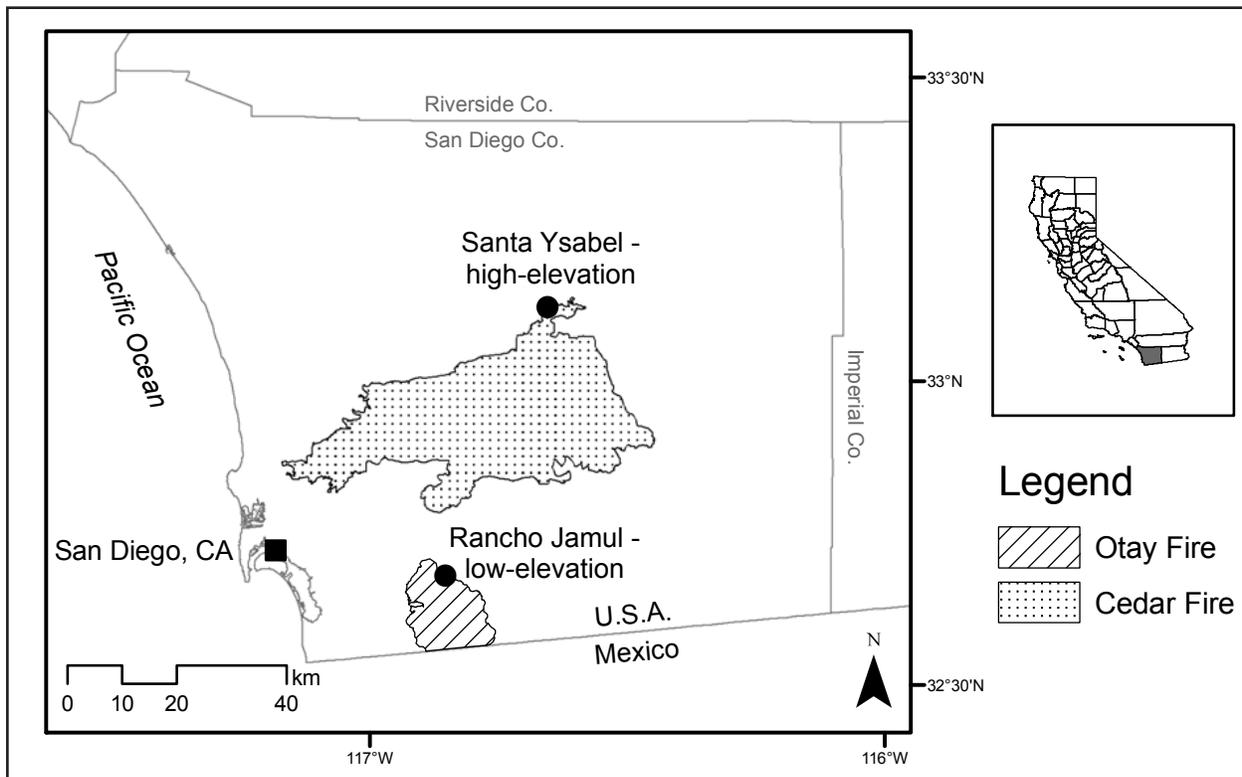
the fires at a low-elevation site and a high-elevation site in each of five vegetation types: (1) chaparral, (2) coastal sage scrub, (3) grassland, (4) oak woodland, and (5) riparian. The primary objective of this study was to examine the short term response of birds to large-scale, patchy wildland fire. The goals were to learn: (1) whether fire affected bird diversity, (2) whether fire affected bird community structure, and (3) if community structure was affected, how did fire influence the relative abundance of individual bird species. We hypothesized that breeding bird species would respond differently across grasslands, shrublands (chaparral and coastal sage scrub), and woodlands (oak woodland and riparian areas). We expected a change in the overall bird community structure in response to the reduction in shrub and tree cover based on previous findings (Lawrence 1966, Stanton 1986, Vreeland and Tietje 2002,

Kirkpatrick *et al.* 2006, Haney *et al.* 2008). We investigated change in community structure by comparing the species diversity, the relative abundance of individuals within each species, and the overall community structure before and after the Cedar and Otay Fires.

## METHODS

### Study Sites and Point Count Stations

Our low-elevation site was Rancho Jamul Ecological Reserve, where our survey points had an average elevation of 275 m (Figure 1). Located in the lower foothills of southwestern San Diego County (California, USA), this site was approximately 30 km from the coast with the prevailing vegetation types of coastal sage scrub (dominated by laurel sumac [*Malosma laurina*], California sagebrush [*Artemisia californica*], and California buckwheat



**Figure 1.** Locations of our two study sites with respect to the two major wildland fires of 2003 in San Diego County, California. Santa Ysabel Open Space Preserve served as the high-elevation site and Rancho Jamul Ecological Reserve the low-elevation site.

[*Eriogonum fasciculatum*]) and non-native grasslands (brome grass [*Bromus* spp.] and wild oats [*Avena* spp.]). The site also included patches of chaparral (chamise [*Adenostoma fasciculatum*], laurel sumac, and California sagebrush), oak woodland (coast live oak [*Quercus agrifolia*] and laurel sumac), and riparian vegetation (willows [*Salix* spp.] and western sycamore [*Platanus racemosa*]). In addition to the natural vegetation, there were extensive, fallow agricultural fields. California Department of Fish and Game manages the 2797 ha ecological reserve (that includes the adjacent Hollenbeck Canyon Wildlife Area). The average July maximum temperature was 29 °C, the average January minimum temperature was 6 °C, and annual precipitation averaged between 28 cm and 31 cm. Southern portions of the Rancho Jamul Ecological Reserve burned during the Otay Fire of 2003. The fire severity level, as measured by the Bare Area Reflectance Classification (USDA 2003a), in most burned areas was 2 on a categorical scale of 1 to 4 with 4 referring to the highest severity. A couple chaparral stands scored a severity level of 3, with no burned areas experiencing a 1 or 4 severity level.

Our high-elevation site was the Santa Ysabel Open Space Preserve, with an average point elevation of 1084 m (Figure 1). This site spans the upper foothills and lower mountains of north-central San Diego County, approximately 60 km from the coast. Oak woodland (characterized by coast live oak and Engelmann's oak [*Quercus engelmannii*]), chaparral (canyon live oak [*Quercus chrysolepis*], California sagebrush, and manzanita [*Arctostaphylos* spp.]), and native and non-native grasslands (needle grass [*Nassella* spp.], wild oats, and brome grass) were the dominant plant communities. Isolated patches of coastal sage scrub (white sage [*Salvia apiana*] often with coast live oak associated) could be found on-site, as well as riparian corridors (coast live oak and western

sycamore) such as Santa Ysabel creek. The County of San Diego Parks and Recreation Department manages this 2188 ha preserve (which includes both the east and west parcels). The average July maximum temperature was 33 °C, the average January minimum temperature was 1 °C, and the annual precipitation averaged 53 cm. Santa Ysabel Open Space Preserve falls within the northeastern extent of the Cedar Fire, which burned approximately half of the east parcel, but none of the west parcel, in 2003 (Figure 1). At our burned survey points, the fire severity level averaged 1.8, ranging between 1 and 3 on a scale of 1 to 4 (USDA 2003b). While none of our survey points experienced high fire severity, several small patches within and adjacent to our survey sites burned at the highest severity level.

All temperature and precipitation values we report here are 30 yr averages (1966-1995; Franklin *et al.* 2001). We used geographic information system (GIS) application to extract these values for each point.

We conducted avian point count surveys at 30 random points per site; we selected these points with a GIS and an existing vegetation map (San Diego Association of Governments 1995). We spaced each point count station  $\geq 250$  m away from any other station. Within both high and low sites, we selected six points in each of the five vegetation types: chaparral, coastal sage scrub, grassland, oak woodland, and riparian. Within each vegetation type, we located three points in areas that burned in the 2003 fires and put the other three in areas outside of the fire perimeter. Across both high- and low-elevation sites, we located a total of 12 replicates within each vegetation type. We considered all of the points to be in mature, unburned vegetation during the pre-fire sampling effort. Each point served as the center of a 100 m radius plot within which we recorded the birds we observed.

We qualitatively and quantitatively assessed the vegetation within the 100 m radius plot around each station following methods similar to those in Ralph *et al.* (1993), Cooper (1999, 2002), and the Point Reyes Bird Observatory Point Count Veggie (Relevé) Protocol (PRBO 2002). We performed an assessment once at each station pre-fire, and once in each of the two post-fire years. At each plot, we recorded the vegetation structure (e.g., percent cover by shrubs and trees, herbs, or open dirt and rock), species composition, maximum vegetation height, percent native cover, substrate type, slope aspect and degree, and hydrology and road presence. Among these metrics, we determined that only the shrub and tree cover, taken as a composite, was suitable for our analysis to investigate whether changes in the vegetation cover resulting from the wildland fires were related to changes in the bird community. We classified vegetation cover into one of six categories: 0 %, 1 % to 4 %, 5 % to 25 %, 26 % to 50 %, 51 % to 75 %, and 76 % to 100 %.

#### *Point Count Surveys*

We surveyed birds between April and July at all stations using standard point counts initiated between 0530 and 1100, with methods and data forms based on those of Ralph *et al.* (1993, 1995). With a few exceptions, we used a 10 min, 100 m fixed-radius point count technique and recorded all birds we observed (visually or audibly) at the point (Appendix I). To standardize the count data for analyses, we used the number of detections per minute for each species. Expecting that the number of detections per minute was not a linear relationship, we ran data simulations and confirmed that our normalization procedure had no effect on the results of our analyses. We conducted pre-fire count surveys at the low-elevation site in 2001 and 2003 and high-elevation surveys in 2002 and 2003 (Appendix

I). We collected post-fire data at both sites in 2005, approximately 18 mo following the fires, and again in 2006.

Variability in detection rates can potentially bias the results of bird point count surveys and, therefore, other researchers recommend applying correction factors to the raw count data to compensate for this bias (Rosenstock *et al.* 2002, MacKenzie and Kendall 2002, Ellingson and Lukas 2003). However, we were not able to apply these methods due to data set limitations such as a lack of distance estimates for each individual in the pre-fire bird point counts. For our analyses, we used the raw count data, which we did not adjust for detection rate variability due to different bird species characteristics or vegetation types or complexity. Just as Hutto and Young (2003) stated, we felt that, despite the suspected biases that may exist, our point count results were sufficient to detect large-scale changes should they occur.

#### *Data Classification and Reduction*

Before we conducted data analyses, we used two different data classification and reduction techniques, depending on the test conducted. Where we conducted univariate tests, we used a before-after reference-impact procedure (BARI) to classify our data (Stewart-Oaten *et al.* 1986, Underwood 1994, Smith 2002). In this process, we classified each data collection point as either a reference point or an impact point. We classified bird points that were outside of the fire perimeter and not impacted by the fires as reference points. Bird points that were within the 2003 fire perimeters and impacted by the fires we called impact points. These classifications also applied to the pre-fire (before) time period when all points were technically unburned and the post-fire (after) time period when impact plots burned either wholly or partially. We used this BARI classification for assessing and

describing the effects of the fires on vegetation, bird diversity, and abundance of individuals within a species.

The multivariate analysis required us to use a slightly different data classification. To have sufficient power to test whether fire had a significant effect on community structure within each vegetation type, it was necessary for us to reclassify and condense the fire classification into only two categories: razed and non-razed. In terms of the BARI classifications described above, all pre-fire surveys (before reference and before impact) plus the post-fire surveys at unburned points (after reference) form the non-razed condition. We called the data collected at the post-fire burned survey points (after impact) the razed samples.

### *Data Analyses*

To analyze our avian species data, we averaged the results from the multiple rounds of count surveys at each point into two samples per point—one before and one after the fires. Within each of these two samples, we averaged the number of detections per species to generate a single abundance index for each species, and we used individual bird count stations as replicates.

Before pooling the 2005 and 2006 post-fire bird data into a single post-fire dataset, we tested whether the vegetation cover significantly changed between these two post-fire years. If there was no change in cover, we combined the bird data for these years into a single post-fire dataset. We converted the six categories of cover into indexed values, one through six, for the analyses. We ran general linear models with two-factor interactions. The independent variables were vegetation type, year, and burn condition (reference vs. impact), with the change in shrub and tree cover composite as the response variable.

We also tested whether the fires significantly affected vegetation cover. Again,

we ran general linear models with two-factor interactions to test for significance. Vegetation type, time period (before and after), and burn condition (reference vs. impact) were the independent variables and the change in shrub and tree cover was the response variable.

To assess changes in the avian community, we first tested whether the fire had an effect on the overall species diversity using the Shannon index (Shannon 1948). We used the numbers of individuals of each species detected per minute, averaged across all visits to a particular survey point before and after the fires, as the base abundance index values. The response variable was the difference between the Shannon index value before and after the fires (i.e., change in diversity), with the survey point as a replicate for a total of 30 replicates. We used a general linear model to test the main effects and interactions of vegetation type, burn condition (reference vs. impact), and the difference between post-fire and pre-fire shrub and tree cover (i.e., change in cover) on changes in bird diversity ([post-fire diversity] – [pre-fire diversity]). We used t-tests of least squares means to investigate any significant interaction effects. We set our level of significance at  $\alpha = 0.10$  for all statistical tests. We checked for deviations from normality of residuals using Shapiro-Wilk and Kolmogorov-Smirnov tests. Data for the univariate analysis were analyzed using SYSTAT (Version 10, SPSS, Inc., Chicago, Illinois, USA) and SAS (Version 9.1, SAS Institute, Cary, North Carolina, USA) statistical software.

Changes in the avian community may not be reflected by a single measure of diversity. Therefore, we also tested for changes in the abundance of individual species and for shifts in the overall bird community structure by using additional multivariate and univariate techniques. We performed multivariate analyses first, using the statistical software PRIMER-E (Version 6, Plymouth, United Kingdom; Clarke 1993). Before analyzing the data with PRIMER-E, we square-root

transformed the data to remove some weight from the most abundant species for a more balanced community analysis and created a Bray-Curtis similarity matrix among all samples (Clarke and Green 1988). To generate the Bray-Curtis similarity matrix, PRIMER-E calculated the percentage of similarity between each sample in the dataset by comparing the species that occurred in each with the frequency we detected each species (Clark and Warwick 2001). PRIMER-E measured the similarity as a percentage, 0 % to 100 %, with a similarity of 0 % indicating that two samples had no species in common. If all of the same species occur in two samples in exactly the same numbers, the similarity between the two samples would be 100 % (Bray and Curtis 1957). Using the Bray-Curtis similarity matrices, we used PRIMER-E to test our hypotheses in a stepwise fashion using two-way crossed analysis of similarity (ANOSIM) tests, a multivariate permutation-based test similar to the analysis of variance (ANOVA) test in univariate statistics (Clark and Green 1988). We expected different vegetation types to support different communities of bird species; therefore, we first tested whether vegetation type significantly predicted bird community structure. If so, we tested the effect of the fire separately for each vegetation type. The test statistic  $R$  from an ANOSIM test reflects the observed differences between groups in comparison to within groups and can range from -1 to 1. An  $R = 1$  indicates complete discrimination among groups, while  $R \leq 0$  indicates no discrimination. PRIMER-E calculates significance by permutation (Clark and Green 1988).

To visualize the results of the similarity matrix, PRIMER-E generates non-metric multidimensional scaling (MDS) plots to ordinate the similarities between the samples (Kruskal 1964). These plots are a two-dimensional representation of the multidimensional similarity between samples, which

can be viewed from multiple angles, from front or back, or rotated and flipped. The only scale in an MDS plot is the relative distance between samples, with similar samples appearing close together and dissimilar samples farther apart (Clark and Warwick 2001). Because PRIMER-E limited the MDS to two dimensions, there was some level of distortion or stress incorporated into the ordination in order to best represent the data. Stress is a measure of the combined deviation from the ideal distance between all samples. Two-dimensional MDS plots with stress values  $\leq 0.1$  are considered to be a good representation of the similarity matrix, while MDS plots with stress  $> 0.3$  are considered to be poor representations of the data (Clark and Warwick 2001). Because we used a condensed data classification to compare our non-razed samples to the razed samples, we carefully reviewed the MDS plots for each subset of samples to ensure that there were no systematic effects of time or burn condition that may cause erroneous test results.

We used the ANOSIM results as an omnibus test for limiting Type 1 error and determining if further analyses were warranted. If the condensed fire classification (razed and non-razed) was not a significant predictor of bird community structure within each vegetation type, we conducted no further tests. If the fire classification was significant, we conducted two additional tests; the first was to look for patterns between the amount of shrub and tree cover and the bird community, and the second was to investigate the responses of individual species. We used the biotic-environmental matching (BioEnv) procedure within PRIMER-E to determine if the total shrub and tree cover was significantly associated with the differences between the bird communities in the razed and non-razed samples (Clarke 1993, Clarke and Ainsworth 1993). The BioEnv procedure generates similarity matrices of the bird community samples by ranking them from high to low

similarity. Similarly, it ranks the percent of shrub and tree cover from high to low values. These are then compared using a weighted Spearman statistic ( $\rho$ ). The test statistic,  $\rho$ , can range between -1 and 1; where 1 indicates complete correlation between the bird community structure and the percentage of shrub and tree cover,  $\rho = 0$  indicates no correlation, and -1 represents complete opposition. BioEnv calculates significance by permutation (Clarke and Ainsworth 1993).

To analyze the relative abundance among individual species in the community, we performed two-tailed t-tests on the net change in abundance to determine if the changes at the impact points were significantly different from the changes at the reference points. After removing rare species to satisfy assumptions of normality, we pooled the variances across species within each vegetation type within each site. We considered a species to be rare if we detected individuals of the species at  $\leq 25\%$  of the samples within the vegetation type at each site. Because there were a large number of species detected across all vegetation types, many of which had low abundance values, we chose to focus the results and discussion on the five most abundant species within each vegetation type both before and after the fires within each elevation (Appendix II). In certain cases, we discussed the more rare species in order to address sensitive or invasive species. However, due to the low detection rates of these rare species, we generally refrained from reporting any statistical findings for these results.

## RESULTS

### *Vegetation*

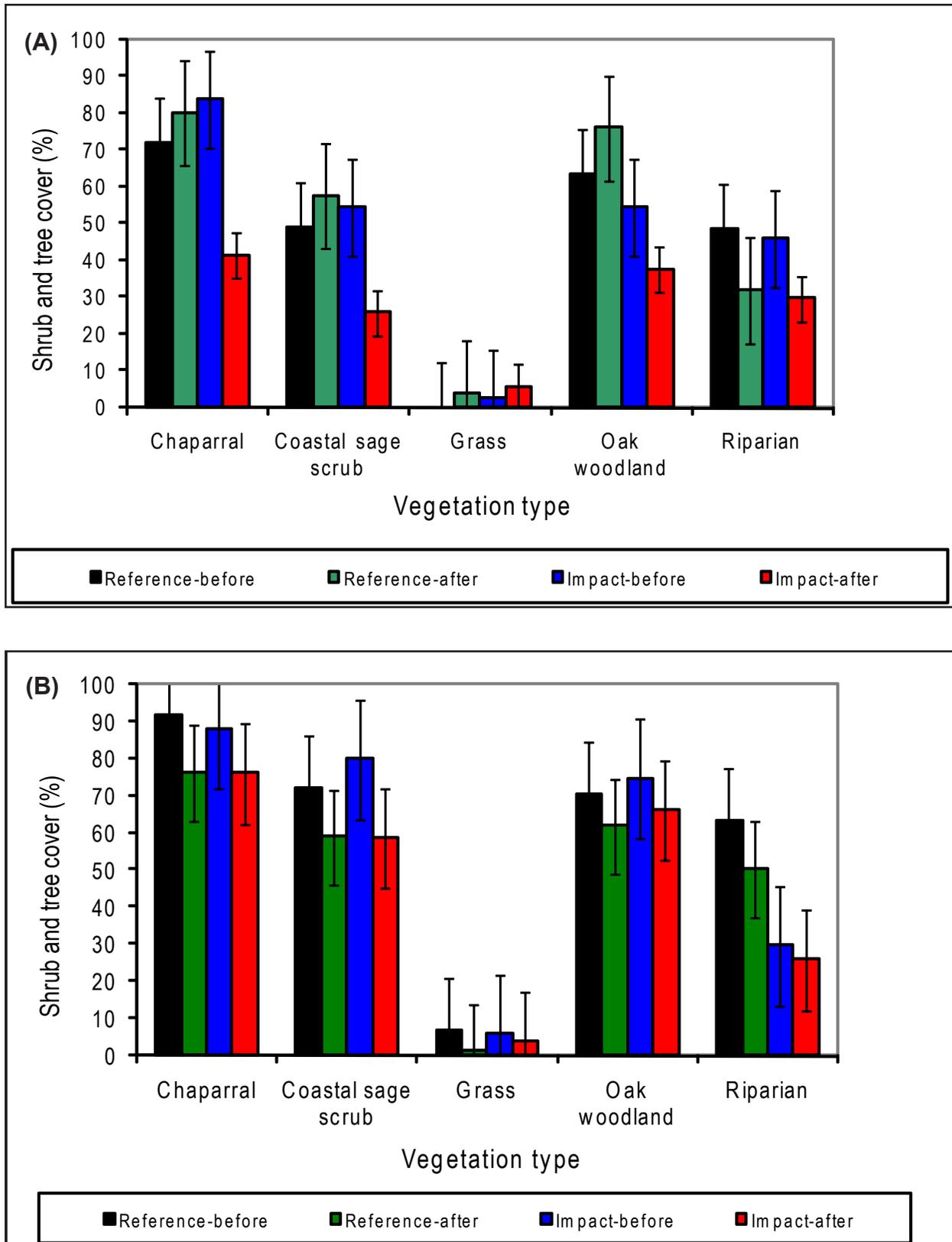
There were no substantial within-site changes in shrub and tree cover during the two post-fire years, 2005 and 2006, as evidenced by year not having a significant main or

interaction effect. Therefore, for the purpose of analyzing the overall effects of fire on bird communities, we combined the 2005 and 2006 data, vegetation cover, and bird point count results into a single post-fire dataset.

The post-fire change in the percent of shrub and tree cover at the reference and impact (burned) points varied by vegetation type and site. At the low-elevation site, impact points in chaparral and coastal sage scrub showed significant decreases in shrub and tree cover ( $t_{40} = -3.03$ ,  $P < 0.05$ , and  $t_{40} = -2.12$ ,  $P < 0.05$ , respectively) after the fire. The reference (unburned) points in the same vegetation types showed no change (Figure 2). In chaparral impact points, shrub and tree cover dropped from an average of 84 % (SE = 10 %) before the Otay Fire down to 41 % (SE = 10 %) after the fire. The cover at impacted coastal sage scrub points decreased from 55 % (SE = 10 %) before the fire to 26 % (SE = 10 %) after the fire. The cover at impact points in grassland vegetation significantly increased from 3 % (SE = 5 %) to 5 % (SE = 5 %) after the fire ( $t_{40} = 3.03$ ,  $P < 0.05$ ). We detected no significant changes in cover in oak woodlands or riparian points in either the reference or impact points at the low-elevation site. At the high-elevation site, we found no significant changes in shrub and tree cover after the fire at any of the five vegetation types. The non-significant but consistent drop in cover in each vegetation type at the high-elevation site (Figure 2) may have been due to extensive cattle activity or a small portion of the 100 m radius plot having been burned (although the majority of the plot remained unburned such that we classified the point as a reference point).

### *Bird Species Diversity and Community Structure*

We detected 2598 individuals of 88 species at the low-elevation site (during 1138 min of counts before the fire and 1200 min post-fire).



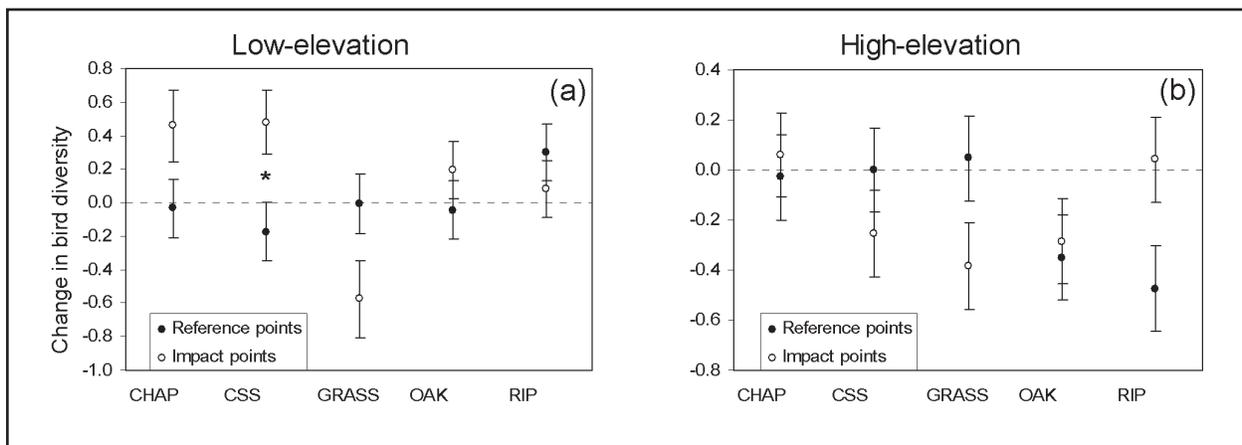
**Figure 2.** Average shrub and tree cover at bird points at (A) the low-elevation site and (B) the high-elevation site in San Diego County, California.

At the high-elevation site we detected 4364 individuals of 89 species (during 1800 min pre-fire and 1200 min post-fire). In total, we detected 6962 individual birds of 107 species during all point counts across both sites. The five most commonly detected species at the low-elevation site across all samples, in descending order, were: California towhee (*Pipilo crissalis*), spotted towhee (*Pipilo maculatus*), wrentit (*Chamaea fasciata*), bushtit (*Psaltriparus minimus*), and mourning dove (*Zenaida macroura*). At the high-elevation site, the five most commonly detected species across all samples were: spotted towhee, lazuli bunting (*Passerina amoena*), ash-throated flycatcher (*Myiarchus cinerascens*), mourning dove, and oak titmouse (*Baeolophus inornatus*). Seventy species occurred at both the high and the low-elevation sites. Seventeen of these species carry special status according to local and federal government wildlife agencies (Appendix II).

Elevation (site) was a significant predictor of the change in bird diversity relative to the fires ( $F_{1,39} = 5.51, P = 0.024$ ); thus, we conducted all further analyses separately by site. At the low-elevation site, change in shrub and tree cover ( $F_{1,19} = 7.26, P = 0.014$ ) and the

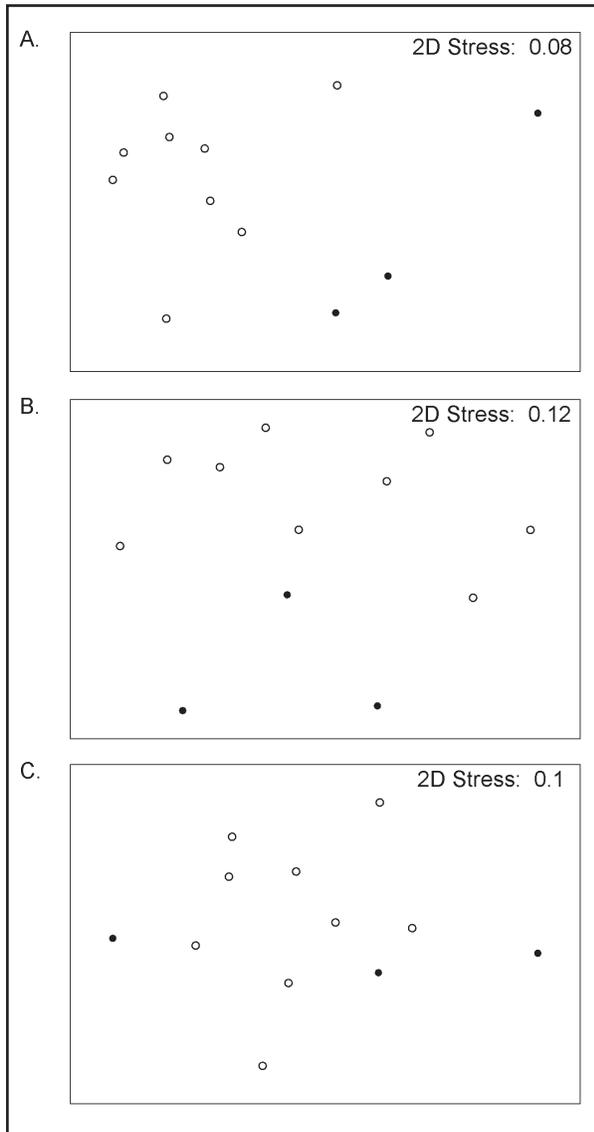
interaction effect of burn condition  $\times$  vegetation type ( $F_{4,19} = 2.63, P = 0.067$ ) were good predictors of post-fire change in bird species diversity. At the high-elevation site, the interaction effect of burn condition  $\times$  vegetation type ( $F_{4,19} = 2.31, P = 0.096$ ) was also significant. The significant interaction effects at both sites indicated that the change in diversity after the fire at impact and reference points was dependent upon vegetation type. Follow-up t-tests on pairs of least squares means showed only one significant change in species diversity. Coastal sage scrub at the low-elevation site experienced an increase in species diversity at the impact points ( $t_4 = -2.36, P = 0.077$ ) (Figure 3). No vegetation type at the high-elevation site showed a significant post-fire change in avian species diversity.

For the low-elevation site, our initial ANOSIM analysis of the pre-fire data revealed that the bird community structure differed among the five vegetation types ( $R = 0.308, P < 0.001$ ). Therefore, we tested the effect of the fire for each vegetation type separately. Bird community structure differed between the razed and non-razed samples in both chaparral ( $R = 0.793, P = 0.005$ ) and coastal sage scrub



**Figure 3.** Least squares mean (plus and minus one standard error displayed) visuals for the change (post-minus pre-fire) in bird Shannon diversity index (both y-axes, but note different scales) after the fire of the impact and reference points for bird point counts in San Diego County, California, USA. Results by vegetation type (x-axes) at the low-elevation site (a), and high-elevation site (b). CHAP = chaparral, CSS = coastal sage scrub, GRASS = grassland, OAK = oak woodland, RIP = riparian. (\* = significantly different)

( $R = 0.254$ ,  $P = 0.082$ ) (Figure 4), but not in the grassland ( $R = -0.031$ ,  $P = 0.532$ ), oak

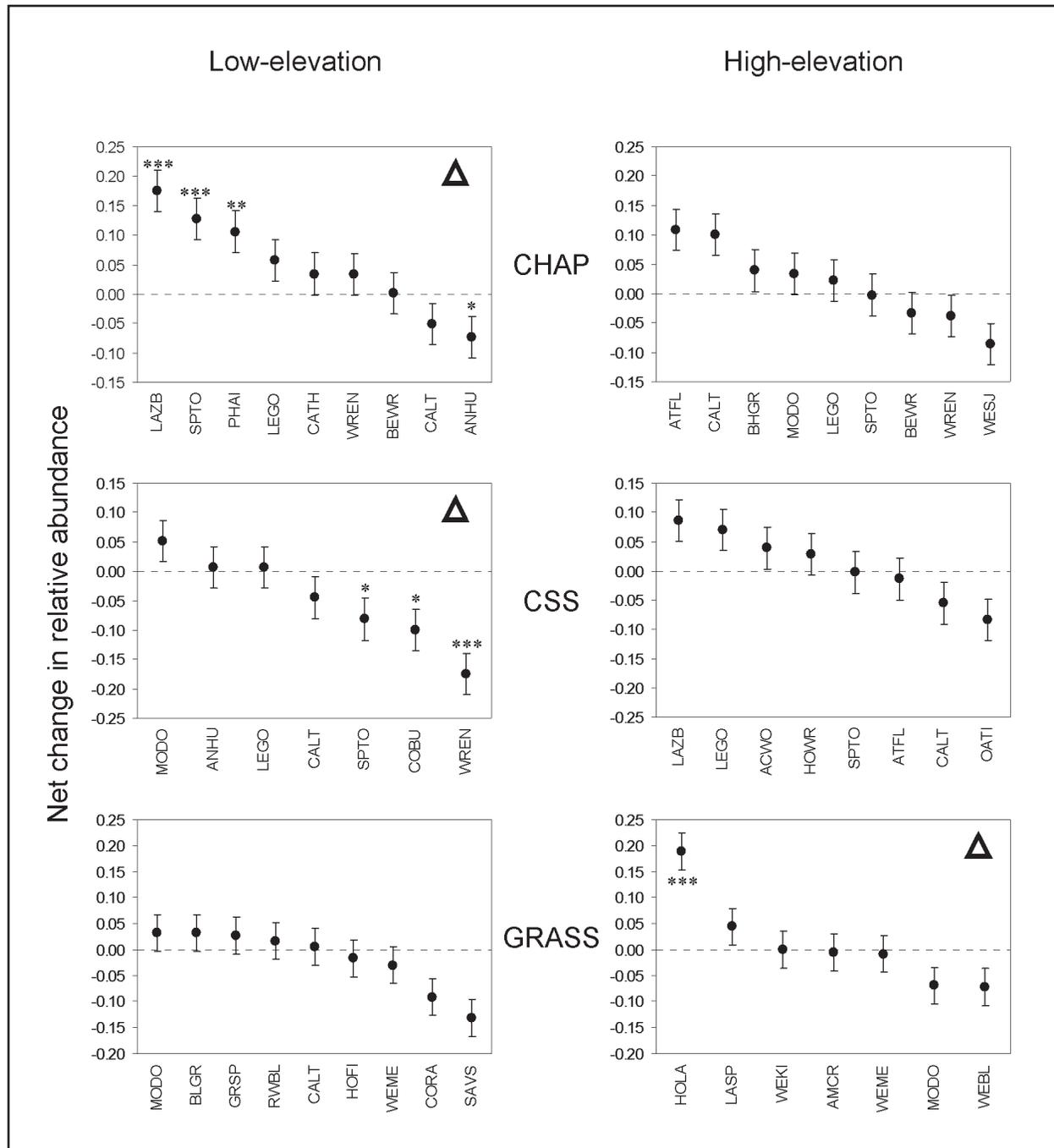


**Figure 4.** Two-dimensional MDS plots for sites and vegetation types where a significant difference was detected between the bird communities within the razed and non-raised samples: (A) the low-elevation chaparral, (B) the low-elevation coastal sage scrub, and (C) high-elevation grassland in San Diego County, California. Each marker represents the bird community from one sample, and each marker's position correlates to the similarity between the sample and those samples around it. The closer two markers are, the more similar are the bird communities at each. Distant markers represent less similarity between samples. Open markers are non-raised samples and solid markers represent razed samples.

woodland ( $R = 0.157$ ,  $P = 0.186$ ), or riparian ( $R = 0.117$ ,  $P = 0.245$ ) vegetation types. The shrub and tree cover associated with each sample was an excellent predictor of the bird community structure in both chaparral (BioEnv:  $\rho = 0.550$ ,  $P = 0.002$ ) and coastal sage scrub ( $\rho = 0.401$ ,  $P = 0.010$ ) at the low-elevation site.

At the high-elevation site, pre-fire bird community structure also differed significantly among vegetation types (ANOSIM:  $R = 0.415$ ,  $P = 0.001$ ). Again, we tested the effect of the fire for each vegetation type separately. Here, community structure differed between non-raised and razed samples in only the grassland vegetation type ( $R = 0.316$ ,  $P = 0.064$ ) (Figure 4). Although the average shrub and tree cover at both the non-raised and razed samples was  $\leq 8\%$ , the BioEnv test indicated that the bird community structure was associated with the amount of shrub and tree cover ( $\rho = 0.322$ ,  $P = 0.011$ ). Bird community structure did not differ between non-raised and razed samples in the chaparral ( $R = 0.039$ ,  $P = 0.450$ ), coastal sage scrub ( $R = -0.031$ ,  $P = 0.559$ ), oak woodland ( $R = 0.206$ ,  $P = 0.177$ ), or riparian ( $R = -0.193$ ,  $P = 0.868$ ) vegetation types.

The set of the five most abundant species in each vegetation type differed before and after the fires and between high- and low-elevation sites (Appendix II). In the low-elevation chaparral, three of the most abundant species before and after the fire (lazuli bunting, spotted towhee, and phainopepla [*Phainopepla nitens*]) significantly increased at the impact points after the fire, while one species (Anna's hummingbird [*Calypte anna*]) significantly decreased, and five others did not change (Figure 5). In contrast, three of the most abundant species in coastal sage scrub (wrentit, bushtit, and spotted towhee) significantly declined at the impact points after the fire, while four others showed no change. In the high-elevation grassland, only the horned lark (*Eremophila alpestris*) significantly increased



**Figure 5.** Least squares means for the most abundant bird species before and the most abundant after the fire (some remained the same), within each vegetation type at both elevations from point counts in San Diego County, California. The estimate (plus and minus one standard error displayed) is the net change in the average relative abundance (number of individuals per minute) for each. \*\*\*  $P < 0.001$ ; \*\*  $P < 0.01$ ; \*  $P < 0.10$  for individual species.  $\Delta$  = the bird community structure differed significantly between non-raized and razed samples within the vegetation type at the reported elevation. Tests for changes in the relative abundance of species were only performed where there was a significant change in the community structure as indicated by the ANOSIM results. CHAP = chaparral, CSS = coastal sage scrub, GRASS = grassland. Refer to Appendix II for bird species code explanations.

after the fire, with no differences in the remaining six species.

## DISCUSSION

Overall, we found relatively few changes in the avian communities studied in the second and third years after the patchy wildland fires at our high- and low-elevation study sites. Where we detected differences at our sites, the percent of vegetative cover present on our plot influenced the patterns in avian community structure that we observed. Within chaparral and coastal sage scrub communities at the low-elevation site, where shrub and tree cover was lower after the Otay Fire, we detected differences within the avian community structure. Concomitantly, bird species diversity increased in the impacted coastal sage scrub. At the high-elevation site, we detected little change in the shrub and tree cover and, likewise, there was little change in the avian community structure or species diversity.

Within the points and vegetation types where we observed no changes in avian species diversity or community structure, there are multiple possible factors contributing to the apparent resiliency of the bird communities, including the availability of nearby unburned refugia, and the generally high mobility of this taxon. The variable nature of these fires resulted in a mosaic of high severity, low severity, and unburned patches of vegetation within the fire perimeter. The nearby and adjacent unburned areas likely served as refugia into which birds that survived the fire could migrate, and then from which birds could re-colonize the post-fire landscape (Whelan 1995). The median distance of impact (burned) points to the perimeter of the fire at the low-elevation site was 959 m (range: 17 m to 2288 m), and 487 m (range: 61 m to 951 m) at the high-elevation site (USDA 2003a, 2003b). For comparison, the reference

(unburned) points had a median distance of 822 m (range: 25 m to 3548 m) from the fire perimeter at the low-elevation site, while the high-elevation unburned points were a median 1350 m (range: 35 m to 4797 m) outside the fire's boundary. Therefore, all but the least vagile birds could have flown  $\leq 2$  km to escape direct mortality associated with fire and potentially set up new territories outside the fire's perimeter. The combination of the patchwork nature of fire on the landscape and the mobility of birds results in the high likelihood that, for many species, their territories include both burned and unburned vegetation. As our results show, the vegetation structure at many of the impacted points was similar to that recorded before the fires, at least with regard to shrub and tree cover. It appears that both of these high- and low-elevation study sites were suitable landscapes post-fire for many of the same bird species as before the fires. Within each vegetation type, we detected several additional notable results.

### *Chaparral*

We observed that bird communities in chaparral had a mixed response to fire. In the impacted low-elevation chaparral, there was a 50 % decrease in the vegetation cover. Our multivariate results show that the percent of shrub and tree cover at each point was associated with the differences in the avian community structure. Although we were able to detect these differences, we observed no significant changes in the overall species diversity. Within the high-elevation chaparral, we detected no change in any of the measured indices. Previous studies investigating diversity or richness in post-fire chaparral have also reported mixed results. For instance, Lawrence (1966) concluded that in bird species displaying a strong affinity for chaparral, the number of individuals was substantially reduced in the three years following a burn in

the Sierra Nevada foothills of California. In contrast, Wirtz (1979, 1982) suggested that numbers of individual birds increased for  $\leq 3$  yr post-fire in burned chaparral areas in California's San Gabriel Mountains. Despite these changes in abundance, he found that avian species richness and the trophic guild structure in burned chaparral were roughly equivalent to those in unburned chaparral.

We observed variable responses by the individual bird species, most likely associated with the life characteristics of each species. Lazuli bunting increased in abundance after the fire in our low-elevation chaparral impact points. Wirtz (1979) showed comparable results; the bunting represented 11 % to 25 % of his sightings in burned chaparral stations. However, he observed fewer buntings in unburned chaparral, reporting them as only occasional sightings (1 % to 10 % of all observations). The bunting is an open-cover specialist, preferring to perch on branches with high visibility and forage over open ground or ground with little shrub and tree cover (Unitt 1984). It follows that the bunting would actively select burned areas over unburned ones (USDA 2007).

The spotted towhee also increased in our low-elevation impacted chaparral, relative to the reference stations. Spotted towhees prefer a large element of shrub and tree cover for both foraging and breeding. This species forages by scratching through leaf litter to find insects and seeds (Unitt 1984). The fire severity level was 2 or 3 at our three chaparral impact points, a level that may have exposed food items for the towhee, while not completely eliminating the necessary cover. Wirtz (1979) reported spotted towhees to be only 1 % to 10 % of the birds observed in both unburned and burned chaparral at 975 m, but that they increased to 11 % to 25 % of his observations in the 1280 m elevation burned chaparral. McClure (1981) recorded the survival of many spotted towhees after a wildland fire in Ventura County,

California, but also noted fire-related beak and foot injuries.

The apparent increase in the abundance of phainopeplas at the impact points in the low-elevation chaparral warrants further mention. The significant result in our data came from observations during six samples before the fire, compared with two after the fire, both at reference points. Our data (Appendix II) revealed that phainopepla detections decreased at both the reference and impact points. However, the decrease within the impact points was substantially smaller than the decrease within the reference points. This indicates a favorable response at the impact points according to our BARI data reduction process. However, our results should be interpreted with caution until substantiated by other studies due to the low detection rates we observed (Appendix II). Although Stanton's (1986) results for phainopepla are from coastal sage scrub, she reported that they preferred unburned vegetation.

Anna's hummingbirds decreased after the fire in low-elevation chaparral in our study. This decrease may be due to individuals shifting territories to neighboring stands of coastal sage scrub or another vegetation community offering more suitable foraging or nesting resources. This decline is unlikely to have been caused by a fire-related decrease in nectar-producing flowers; many perennial and annual herbs with showy flowers bloomed during the time over which we conducted our post-fire surveys. In another study, Wirtz (1979) found the Anna's hummingbird to exhibit no strong response to fire; it was 1 % to 10 % of all the birds detected both before and after a fire in chaparral at 975 m and 1280 m elevation sites.

### *Coastal Sage Scrub*

Many of the large-scale patterns that we detected in chaparral also occurred in coastal

sage scrub. As in the low-elevation chaparral, the impacted coastal sage scrub from the same site also experienced a 50 % decline in shrub and tree cover. Similarly, our multivariate results showed that the percentage of shrub and tree cover influenced the coastal sage scrub avian community structure. Unlike the nearby chaparral, we found a post-fire increase in bird diversity in low-elevation impacted coastal sage scrub. Stanton (1986) observed bird responses to fire two and three years post-fire at a 300 m elevation site in Los Angeles County, California. Although this was similar to our low-elevation site, she concluded that species diversity in the unburned coastal sage scrub was generally higher than in burned areas, the opposite of our findings. However, differences in our sampling protocols and study sites may explain the inconsistencies between our and Stanton's (1986) findings.

The species that showed significant declines in low-elevation coastal sage scrub, such as the spotted towhee, wrentit, and bushtit, all generally prefer vegetation communities with high cover. The post-fire reduction in cover in the burned chaparral possibly led this vegetation type to appear and function similar to intact coastal sage scrub; the burned chaparral could now provide perches with high visibility and, therefore, lure in birds that typically use open scrubby areas. One species in our study, the spotted towhee, decreased in impacted coastal sage scrub, but increased in impacted chaparral. These vegetation communities often border each other, so it is reasonable that spotted towhees surviving the fire moved from impacted coastal sage scrub to impacted chaparral to satisfy their habitat requirements. Stanton (1986) also observed significantly fewer spotted towhees in burned coastal sage scrub.

In our study, the wrentit decreased in abundance after the fire in the low-elevation coastal sage scrub. Stanton (1986) observed 125 wrentits in unburned coastal sage scrub,

but none in burned areas, exhibiting a highly significant preference for the former. The wrentit depends on medium to high levels of cover for foraging and nesting (Unitt 1984). As our results showed, fire typically reduces shrub cover in coastal sage scrub. This vegetation type is no longer suitable for wrentits and we would not expect them to persist in a burned patch within the first few years after a fire. Likewise, the bushtit decreased after fire in our study. Stanton (1986) also found bushtits to prefer unburned areas. The bushtit shares many life history traits and requirements with the wrentit and, likewise, we would not expect it to prosper in areas recently denuded of shrub or tree cover.

We detected no change in species diversity or community structure at the high-elevation coastal sage scrub points. Since our high-elevation site had limited, isolated patches of coastal sage scrub that showed no post-fire changes in shrub and tree cover, it is not surprising that we did not see changes in birds relative to the fire.

### *Grassland*

The low-elevation grasslands contained predominantly non-native annual grasses before the fire, most of which recover and even expand quite readily following fire (Lawrence 1966, Smith 2000). Therefore, the grasslands at this site after the fire may have become very similar in structure and species composition to the grasslands before the fire, particularly by the time our post-fire point counts started, 18 mo after the fire. In contrast, the high-elevation grasslands contained a considerably greater amount (both number of individuals and coverage area) of native bunchgrasses. These grasses may take longer to reestablish and are also often outcompeted by non-native grasses and forbs in post-fire situations (Smith 2000). Therefore, the change in bird community structure that occurred in the high-elevation

grassland post-fire, but not at the low-elevation grassland, perhaps reflected a shift in the structure and composition of the vegetation—a shift that was not as pronounced at the low-elevation site.

In our high-elevation grassland, the horned lark appeared to increase after the fire. This species commonly flocks on barren grasslands or other open areas and is known to breed and over-winter in this region (Unitt 1984). The significant signal in our data may be biased by a single observation of a flock of 17 horned larks during one post-fire count at an impact point. Excluding this flock, the significant difference would likely disappear, with 11 individuals observed pre-fire in grassland points and 13 post-fire. Removal of this point did not change the results of our multivariate tests and there was still a significant overall difference in the avian community structure within the high-elevation grassland.

#### *Oak Woodland and Riparian*

Neither oak woodland nor riparian areas exhibited change in shrub and tree cover, avian diversity, or community structure after the fire. Similarly, in Lawrence's (1966) surveys of oak woodland birds, he found no change or only a slight increase in the individual abundance of birds after a fire. Vreeland and Tietje (2002) found no difference in the abundance of individuals after prescribed fire in oak woodlands in northern San Luis Obispo County, California. Given the high levels of diversity known to inhabit both oak woodland and riparian bird communities, more intensive, targeted surveys, possibly with a smaller plot radius, may be required to accurately measure the fire responses of birds obligated to these vegetation types.

#### *Conservation Implications*

Although our study was not specifically designed to address the issues of sensitive or

invasive species, we documented several during our surveys whose presence may have conservation implications (Appendix II). Raptors such as the white-tailed kite (*Elanus leucurus*) and golden eagle (*Aquila chrysaetos*) used both our high and low-elevation sites, most likely due to the large open space provided within the boundaries of these preserved lands. Our incidental observations of raptors hunting at these sites increased following the burns, likely because their prey were more exposed in the burned landscapes (Lawrence 1966, Whelan 1995, Shaffer and Laudenslayer 2006).

Because of the extensive loss of natural riparian corridors in the southwestern United States (RHJV 2004), the riparian vegetation that is protected within Rancho Jamul Ecological Reserve and Santa Ysabel Open Space Preserve may serve as refugia for species closely associated with this vegetation type, such as the least Bell's vireo (*Vireo bellii*), yellow warbler (*Dendroica petechia*), and yellow-breasted chat (*Icteria virens*) (Unitt 1984), all of which we detected in low numbers. Chaparral and coastal sage scrub shrublands like those present at these two study sites provide habitat for many species of interest. We detected some shrubland dependent species, such as the California gnatcatcher (*Polioptila californica*), rufous-crowned sparrow (*Aimophila ruficeps*), and Bell's sage sparrow (*Amphispiza belli*) (Unitt 1984), both before and after the fires (Appendix II) at both study sites. While our detection rates for many of these species were too low to allow insight into their responses to fire, the continued presence of these species may be attributed, in part, to the value of these preserved lands.

Invasive species like the brown-headed cowbird (*Molothrus ater*) and European starling (*Sturnus vulgaris*) were widespread on both sites pre-fire and post-fire (Appendix II). We detected brown-headed cowbirds within only one of the five vegetation types at the

low-elevation site before the fires (oak woodland). However, after the fires, we detected them in all vegetation types except chaparral. At the high-elevation site, the brown-headed cowbird was nearly ubiquitous and observed in all vegetation types both before and after the Cedar Fire, with the exception of the post-fire coastal sage scrub. While the results from our low-elevation site indicated a positive response to fire for the brown-headed cowbird, work by Stanton (1986) showed no preference. However, Powell's (2008) efforts in a tall grass prairie in northeastern Kansas indicated a negative response to fire. We found the European starling within two vegetation types pre-fire at the low-elevation site compared to post-fire, where it was only absent from the chaparral points. In our study, the relative abundance

rates for these invasive species were insufficient to determine if their numbers increased or decreased significantly in response to wildland fire. However, the continued persistence of invasive species may eventually require control efforts to safeguard sensitive species such as the western bluebird (*Sialia mexicana*) and least Bell's vireo.

In the face of the large-scale wildland fires of 2003, these preserved lands appear to have successfully maintained a consistent level of avifauna species diversity. San Diego County experienced a repeat of these large-scale fires in the fall of 2007. The Witch Creek, Poomacha, and Harris Fires, which burned or re-burned portions of the study sites described here, may warrant further study to build on the body of knowledge present herein.

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