Point Blue Report

Monitoring the Effects of Power Fire Herbicide Treatments on Complex Early Seral Forest Birds

Final Report to the Eldorado National Forest

October 2021

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Conservation science for a healthy planet

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Acknowledgements

We'd like to thank the Eldorado National Forest and the Amador Ranger District for financial and logistical support of this work, in particular Robin Wall, Chuck Loffland, Rick Hopson and Becky Estes. We are indebted to our 2019 and 2020 field crew for their dedication and hard work, including Wyatt Hersey, Joel Donohue, Jaime McConachie, and John Garrett; and to the 2014-16 field crew. Thanks to Helen Loffland for helping with GIS work and the ACCG Monitoring Work Group for many insightful discussions related to the Power Fire area.

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Cover photos*: Photos by A. Fogg and Wikimedia Commons (L-R) showing whitethorn ceanothus treated with herbicides in the Power Fire and a female MacGillivray's Warbler, common in older burned areas.*

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INTRODUCTION

After nearly a century of successful fire suppression (Calkin et al. 2005), the subsequent densification of Sierra Nevada forests and accumulation of fuels (Sugihara et al. 2006) has led to increasingly large and severe wildfires across the range (Miller and Safford 2012; Steel et al. 2015). With the important role of fire as a primary driver of ecosystem structure and function, there is a substantial need to understand the value of habitats created and altered by wildfire and how post-fire habitats are used by the unique avian community that occupy them. Management actions in post-fire landscapes affect the forest composition and structure that could persist for decades to centuries (Lindenmayer and Noss 2006, Swanson et al. 2011). Thus, it is prudent to carefully consider the species using post-fire habitats under different management prescriptions, both in the short- and long-term. Reforestation practices have also come under increased scrutiny due to increased frequency and severity of fire and drought events that directly affect post-fire habitat (North et al. 2019) and reforestation objectives that have historically reduced the extent of hardwood, riparian, and early successional habitats (White and Long 2019).

In January 2017, the Amador Ranger District proposed management actions in the 2004 Power Fire primarily to re-establish a fire resilient forested landscape (USDA Forest Service 2017). These actions include mechanically clearing vegetation competing with young conifers, applying herbicides to species competing with conifers, and planting conifer seedlings. In this report, we focus on the effects of the herbicide treatments on the early seral bird community. These treatments were largely targeted applications of glyphosate and triclopyr to deerbrush (*Ceanothus integerrimus*) in previously chainsaw-released areas, and herbicide-only treatments that were dominated by whitethorn ceanothus (*Ceanothus cordulatus*) and bear clover (*Chamaebatia foliolosa*).

Herbicides are a common tool in intensive forestry to control vegetation that competes with desirable conifer species (Shepard et al. 2004, Wagner et al. 2004), thereby increasing tree growth and survival to establish forested conditions more rapidly. These treatments have the potential to inhibit, degrade, or shorten early seral vegetation stages (Wagner et al. 2004). Complex early seral forest (hereafter "CESF") has been identified as an important element of western forests (Franklin et al. 2008, Betts et al. 2010, Swanson et al. 2011, Kwit et al. 2014). Several studies have shown declines of CESF-associated birds in clearcuts and conifer plantations impacted by herbicides (Morrison and Meslow 1983, Betts et al. 2013, Kroll et al. 2017), however studies are lacking specifically from managed post-fire CESF and the Sierra Nevada ecosystem.

To help inform a science-based approach to post-fire habitat management for wildlife we investigated the effects of reforestation herbicide treatments on the CESF avian community in the Power Fire. We modeled avian species abundance changes over time in response to the changes in habitat structure that occurred following salvage and reforestation treatments, with special focus on the effects of herbicide use. Using the fitted abundance estimates at a large set of treated and control locations within the Power Fire perimeter, we assessed the significance and magnitude of treatment on CESF bird abundance and richness and explored the mechanism of herbicide-induced vegetation cover loss on these species. This report describes our study's methods, summarizes avian and vegetation responses to the herbicide treatments, and identifies management recommendations for using post-fire management techniques with a special focus on the effects of herbicide treatments.

STUDY AREA AND METHODS

The Power Fire burned 17,005 acres in October 2004 on the Amador Ranger District of the Eldorado National Forest (ENF), located in the central Sierra Nevada Mountains of California. The proportion that burned at high severity (75-100% reduction in canopy cover) was 38%. Approximately 13,600 acres of the fire was on the ENF (~5200 acres burned at high severity). It was human-ignited and burned predominantly on the south-facing side of the Mokelumne River Canyon. Pre-fire forest structure and composition was moderate to densely stocked ponderosa pine and Sierra mixed conifer. The elevations of avian monitoring locations in Power Fire ranged from 1120 – 2016 m (mean = 1611m; N = 148), roughly matching the elevation range of the entire fire area.

Sampling Design

Survey locations were originally established in 2014 within the Power Fire perimeter (Figure 1) as part of a larger study examining bird-habitat relationships in older fires (Fogg et al. 2017). We selected avian sampling stations from a previously established vegetation sampling grid within the fire (Welch and Safford 2010, Richter and Safford 2016). Transects were typically comprised of 10 points made up of two parallel five-point sub-transects, placed at a diagonal along the vegetation plot grids making point count locations approximately 283m apart. Any given point in a transect was at least 500m from points in other transects. Transects were limited to Forest Service land, slopes with a maximum of 35 degrees, and did not require any major stream crossings. In total, 148 points on 15 transects were surveyed in Power Fire during 2014-2016 bird breeding seasons. For 2019, we removed one transect (PW04) due to its remote location and safety concerns (large volume of decaying snags and logs), thus reducing our sample size to 138 points on 14 transects. In 2019, we also could not visit 7 points within one transect (PW09) due to proximity to marijuana contamination sites.

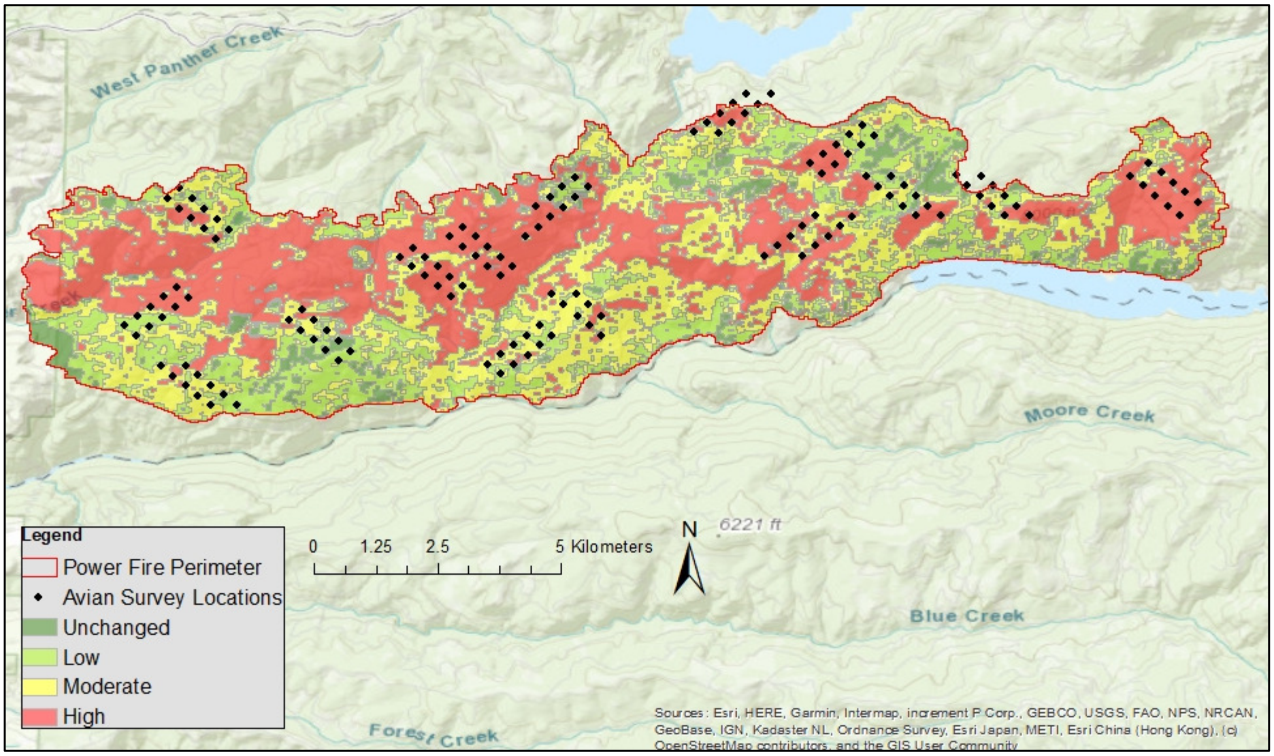


Figure 1. Avian survey locations (N = 148 points) overlaid on a burn severity map for the Power Fire.

Reforestation treatments

The Power Fire Reforestation project (USDA Forest Service 2017) sought to reforest areas that burned at high severity and had low amounts of conifer regeneration, including plantations established post-fire as well as areas with naturally-occurring regeneration. Treatments, proposed across approximately 3500 acres (67% of the high severity burned area on ENF land), included manual herbicide spraying (primarily glyphosate and triclopyr to control competing vegetation [shrubs, grasses, bear clover]); clearing deerbrush with chainsaws (material was left on the ground) and following up with herbicide applications; and bulldozing and piling competing vegetation using heavy machinery and then replanting with conifer seedlings (Figure 2). Herbicide spraying occurred primarily during late spring and summer 2018 with a smaller area treated in 2019, during the height of bird nesting season. Bulldozers were also used to push vegetation into piles and later burned; this was followed by replanting conifers in most areas (N = 6 points with 5-100% of the area within 50m treated). We compiled the treatment history using the Region 5 Forest Activity Tracking System (FACTS) database (available online at http://www.fs.usda.gov/detail/r5/landmanagement/gis) and through ground-truthing surveys where a field technician estimated the area treated within 50m of the survey point center and type of treatment, relying on evidence of dead or removed shrubs. In areas dominated by deerbrush, the chainsaw treatments took place during summer-fall 2018 (N = 9 points with 10-100% of the area treated within 50m) and were then followed by herbicide applications during summer 2018 or 2019. Herbicide-only treatments, primarily in whitethorn and bear clover-dominated areas, took place at N = 22 points with 20-100% of the area within 50m treated during 2018.

We included in our analysis a set of control locations that burned at moderate or high severity and received no treatment within 100m of the point center determined using ground-truthed surveys (N = 53). We limited the treated locations to only those with herbicide applications (N = 31) and controls with no treatments (N = 53) as sample sizes within other treatment categories were too small for meaningful statistical analysis (N < 10). The 84 locations included in the herbicide effects analysis were surveyed 4-5 times from 2014-2020, and treatments occurred primarily in 2018 except at 5 points that had deerbrush cut via chainsaw but was not treated with herbicide until after we collected 2019 bird data. Thus 2019 data represents one year post-treatment and the majority of 2020 data is two years post-treatment. This leaves a final sample of N = 358 point-year survey events in untreated locations including both controls (N = 260) and pre-treatment (N = 98), and N = 52 point-year survey events in post-herbicide-treatment locations.

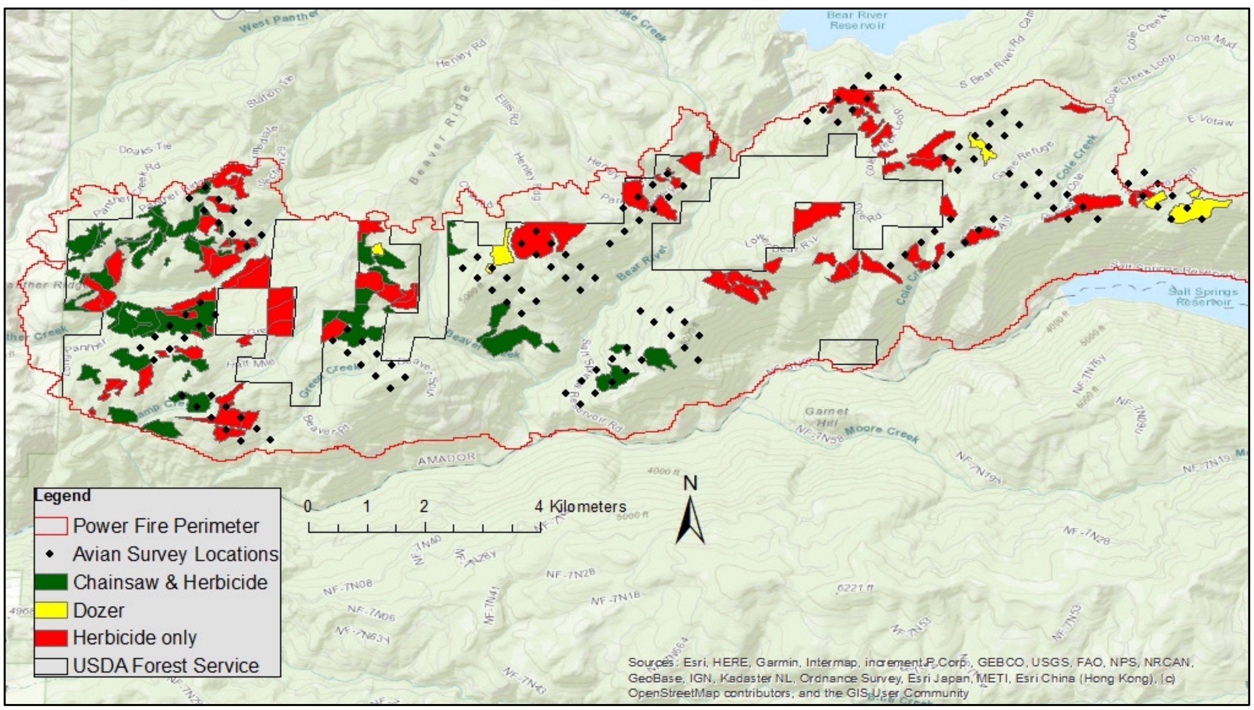


Figure 2. Avian survey locations within the Power Fire area and 2018 reforestation treatments. Chainsaw treatment was the manual cutting of deerbrush and leaving it on the ground. These areas were then later treated with herbicides, in addition to areas only treated with herbicides. Dozer treatment was clearing of the majority vegetation using a dozer and replanting with conifer seedlings (not analyzed). Treatments primarily occurred during 2018 with some follow-up herbicide spraying in 2019 at chainsaw points. At a small number of points, our field crews documented treatments which were not accounted for in the FACTS layer and thus are not accounted for in this figure.

Survey protocols

Experienced observers conducted standardized five-minute exact-distance point counts at each point count station (Ralph et al. 1995). With the aid of rangefinders, surveyors estimated the exact distance to each individual bird. The initial detection cue (song, visual, or call) for each individual was also recorded. Counts began around local sunrise, were completed within four hours, and did not occur in inclement weather. Surveyors received two weeks of training to identify birds and estimate distances and passed a double-observer field test. The majority of transects were visited twice during the peak of the breeding season from mid-May through the end of June during 2014-2016, but were only visited once or twice in 2019 and only once in 2020 due to timing of ongoing herbicide treatments and staffing shortages related to the Covid-19 pandemic.

Vegetation data was collected at all point count locations during July 2014-2016 (pre-treatment) and 2019-2020 (post-treatment). We measured vegetation characteristics within a 50m radius plot centered at each point count station following a modified version of the relevé protocol outlined in Ralph et al. (1993). On these plots, we measured shrub cover, live tree cover, herbaceous cover, as well as the relative cover of each species in the shrub and tree layers (<5m and >5m, respectively). We also measured basal area of live trees and snags using a 10-factor basal area key at five fixed locations in each plot. Fifty five percent of the herbicide treated sample had also been salvaged-logged compared to 30% of the control sample. We summarize vegetation differences between the control and treated samples by plotting all data across all years within each sample with boxplots showing upper and lower quartiles in the box and median values as a solid line.

Analysis

To determine the effects of herbicide treatments, we limited our analyses of bird data to a subset of the species encountered. We *a priori* selected 10 species with sufficient sample sizes in our dataset, and based on the literature and our expert knowledge are associates of the elements of CESF being modified by reforestation treatments. Those species include Mountain Quail (*Oreortyx pictus*), Dusky Flycatcher (*Empidonax oberholseri*), House Wren (*Troglodytes aedon*), Fox Sparrow (*Passerella iliaca*), Spotted Towhee (*Pipilo maculatus*), Green-tailed Towhee (*Pipilo chlorurus*), Nashville Warbler (*Oreothlypis ruficapilla*), Yellow Warbler (*Setophaga petechia*), MacGillivray's Warbler (*Geolypis tolmiei*) and Lazuli Bunting (*Passerina amoena*).

We compiled all bird point count data within 75m of the plot center to generate a dataset of detections of individuals that are near our vegetation assessments (50m radius) while not reducing our bird detections too drastically (detectability for most species begins to decline at approximately 50-75m). We fit generalized linear mixed-effects models (GLMM) including a suite of fixed and random effects to account for topographic, vegetation, and treatment influences on individual species abundance. In these abundance models we included random effects for survey location and year, in addition to fixed effects for elevation, live tree basal area, live shrub species cover, dead shrub species cover, and live <5m tall conifer cover, plus an offset to account for number of visits. To account for treatment effects, we included a binary covariate for treated/untreated (untreated includes both control locations and pre-treatment locations), and an interaction with salvage/unsalvaged. One species (MacGillivray's Warbler) had a non-fitting model with the treatment*salvage interaction term, so we fit the model for that species without the interaction.

We report model coefficients and interpret significant variable effects to describe the mechanisms that drive species abundances across the sample, with a focus on the effects of treatment and vegetation structure. We then plotted the fitted abundance predictions at each point-year sampling unit summarized as average abundance across all points within the treated and control sub-samples to show average and standard error of abundance by year for each species individually, as well as for all CESF species combined. Species richness was estimated from the model-fitted abundance estimates by considering each species as present at a point-year combination if the fitted abundance value was 0.5 or higher, and we then calculated average CESF species richness per point across all points within the control and treated sub-samples.

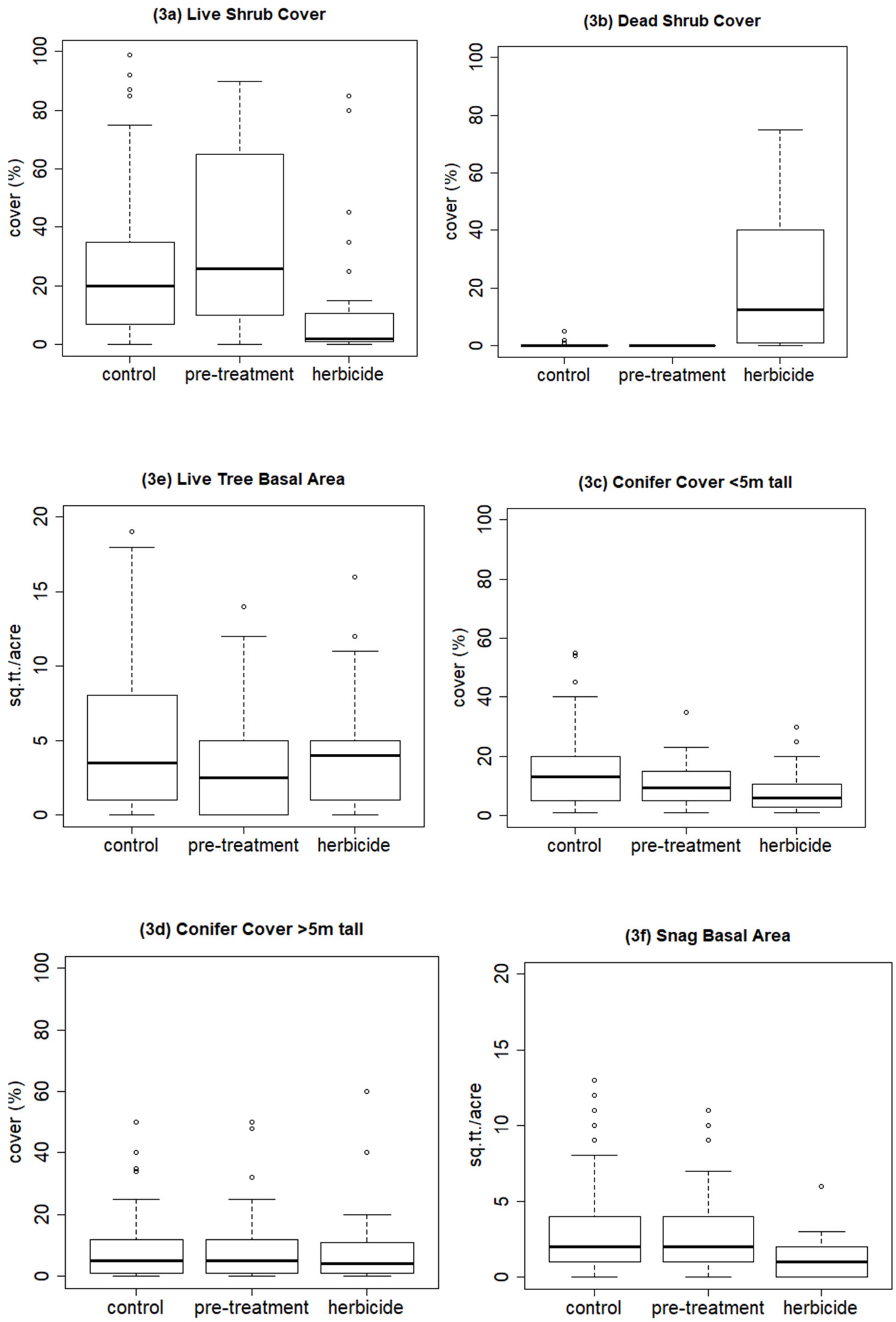
To evaluate the significance of differences between species or CESF abundance and richness across the control vs. treatment sub-samples and through time, we used a two-way ANOVA including time (year) and treatment block (control vs. herbicide sub-samples) and an interaction between those variables. We expected that the time variable would be significant for most species given the strong effects of climate/weather patterns during our study period, and if the treatment variable was significant, we interpreted that as a difference in abundance across sub-samples due to selection of survey locations, not treatment effects. The effect of interest was the treatment*time interaction, which when significant at $P < 0.05$, we interpreted that result as a significant effect of herbicide application on individual species or guild abundance, or species richness. We plot results for each species individually, as well as for guild abundance and richness, and interpret any patterns we found to generate management recommendations.

RESULTS

Treated Vs. Untreated Vegetation

Herbicide treatments resulted in shrub removal in deerbrush ceanothus areas or high shrub mortality where whitethorn ceanothus or bear clover were the dominant species (Figure 3a), however the shrub structure was still present at many of these locations and apparent within our dead shrub measurements (Figure 3b). Treatments had smaller or no effect on young conifer cover (<5m tall; Figure 3c), tree cover (Figure 3d), and live tree basal area (Figure 3e). Dead tree basal area (Figure 3f) was higher in control locations likely reflecting the smaller proportion of salvaged locations.

Figure 3. Vegetation data plotted in control vs. treated locations from pre-treatment (2014-2016) and post-treatment (2019-2020) surveys. Control data from both the pre- and post-treatment periods are combined.



Bird response to herbicide treatments

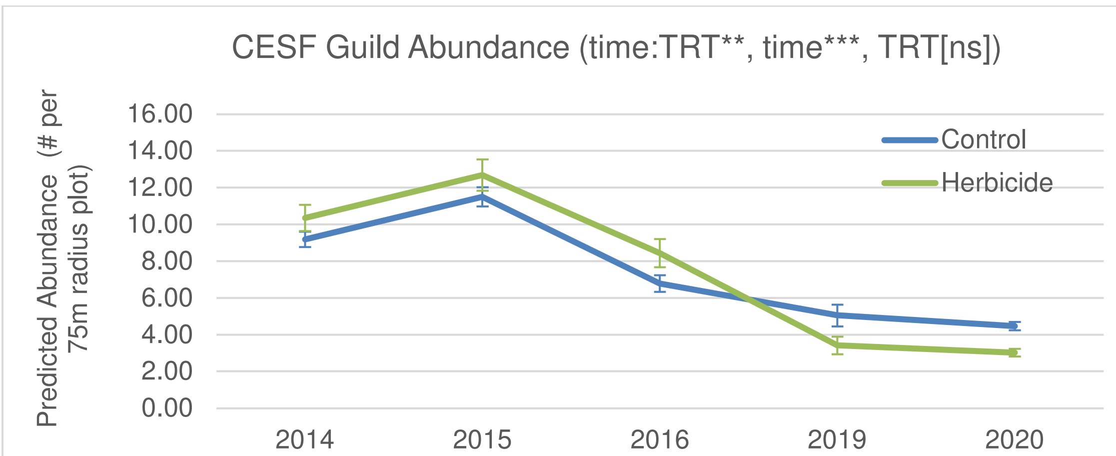
Only two of the ten CESF species (Fox Sparrow and House Wren) had significant negative responses to the herbicide treatment abundance model covariate, while seven of the eight remaining species had non-significant negative responses (Table 1). Shrub cover was a strong positive influence for five CESF species and marginally significant for one more, while live basal area was significant for six species and split evenly between positive and negative (three each). Dead shrub and young conifer cover were significant for only one and two species, respectively. One species had a significant positive response to salvage (Green-tailed Towhee), while two additional species had marginally significant positive responses (Dusky Flycatcher and Mountain Quail), and two negative (House Wren and Nashville Warbler). The treated*salvage interaction effect was significant for none of the species, indicating that the treatment effect is neither stronger nor weaker for CESF species in salvaged locations.

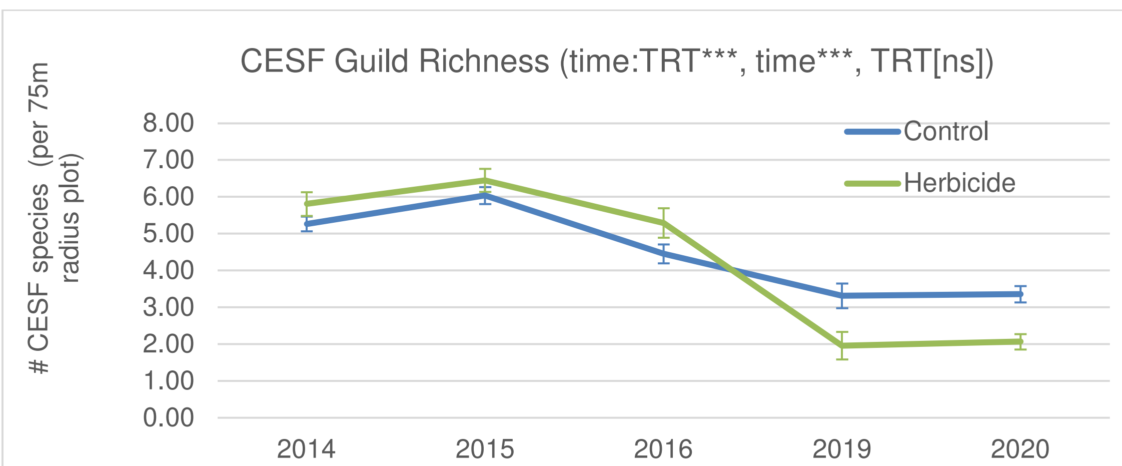
Table 1. Generalized linear mixed effects model coefficients for all ten CESF species. Model coefficients with standard errors in parentheses are bolded for significant effects (p<0.05), or with an asterisk for marginally significant effects (p<0.10).

	Intercept	treated	salvaged	elevation	live basal area	shrub cover	dead shrub cover	young conifer cover	treated: salvaged
Fox Sparrow	**-2.28 (0.16)**	**-1.32 (0.56)**	0.26 (0.20)	**0.80 (0.09)**	**0.21 (0.09)**	**0.49 (0.08)**	**0.31 (0.10)**	0.07 (0.09)	0.51 (0.61)
House Wren	**-1.57 (0.12)**	**-1.08 (0.48)**	-0.28* (0.17)	**-0.33 (0.09)**	**-0.56 (0.10)**	-0.01 (0.07)	0.10 (0.12)	**0.16 (0.07)**	0.51 (0.59)
Spotted Towhee	**-1.75 (0.14)**	-0.17 (0.37)	-0.01 (0.15)	**-0.41 (0.07)**	-0.05 (0.07)	**0.18 (0.06)**	-0.05 (0.12)	-0.02 (0.07)	-0.04 (0.46)
Lazuli Bunting	**-1.82 (0.17)**	-0.25 (0.37)	0.12 (0.13)	**-0.28 (0.07)**	**-0.45 (0.08)**	0.09 (0.06)	-0.15 (0.12)	-0.02 (0.06)	0.02 (0.46)
Nashville Warbler	**-1.89 (0.20)**	-0.23 (0.47)	-0.38* (0.20)	-0.03 (0.10)	-0.13 (0.10)	0.05 (0.09)	-0.09 (0.13)	0.09 (0.09)	0.58 (0.54)
Green-tailed Towhee	**-2.59 (0.17)**	0.16 (0.43)	**0.39 (0.20)**	**0.86 (0.10)**	**-0.21 (0.10)**	0.07 (0.09)	0.09 (0.10)	0.09 (0.09)	-0.60 (0.51)
MacGillivray's Warbler	**-2.47 (0.17)**	-0.74 (0.56)	-0.21 (0.23)	0.20* (0.11)	0.03 (0.10)	**0.33 (0.10)**	-0.09 (0.18)	-0.04 (0.11)	NA (NA)
Dusky Flycatcher	**-3.17 (0.26)**	-0.58 (0.65)	0.51* (0.28)	**0.55 (0.12)**	**0.29 (0.12)**	**0.35 (0.11)**	0.18 (0.12)	**0.25 (0.12)**	0.59 (0.73)
Yellow Warbler	**-3.79 (0.35)**	-0.42 (0.84)	0.19 (0.39)	-0.10 (0.18)	0.01 (0.18)	**0.77 (0.16)**	-0.15 (0.46)	-0.11 (0.20)	-0.80 (1.30)
Mountain Quail	**-4.08 (0.46)**	-0.63 (0.81)	0.72* (0.37)	**0.39 (0.15)**	**0.34 (0.16)**	0.28* (0.15)	-0.02 (0.24)	-0.10 (0.18)	-0.03 (0.96)

ANOVA tests indicated that both total abundance and richness of the CESF guild were higher in control than post-treatment locations, while pre-treatment abundance and richness were slightly higher than in control locations, and results indicated that these patterns were highly significant (Figure 4). Prior to treatments (2014-2016), abundance within pre-treatment locations was 14.9% higher than in control locations (10.49 versus 9.13 individuals per 75m radius plot). However, after treatments (2019-2020) abundance within control locations was 47.8% higher than within post-treatment locations (4.76 vs 3.22 individuals per 75m radius plot). Similarly, CESF richness was 11.4% higher in the pre-treatment locations than control (5.85 vs 5.25 species per 75m plot). Following treatments species richness was 66.2% higher in the control locations (3.34 vs. 2.01 species per 75m radius plot).

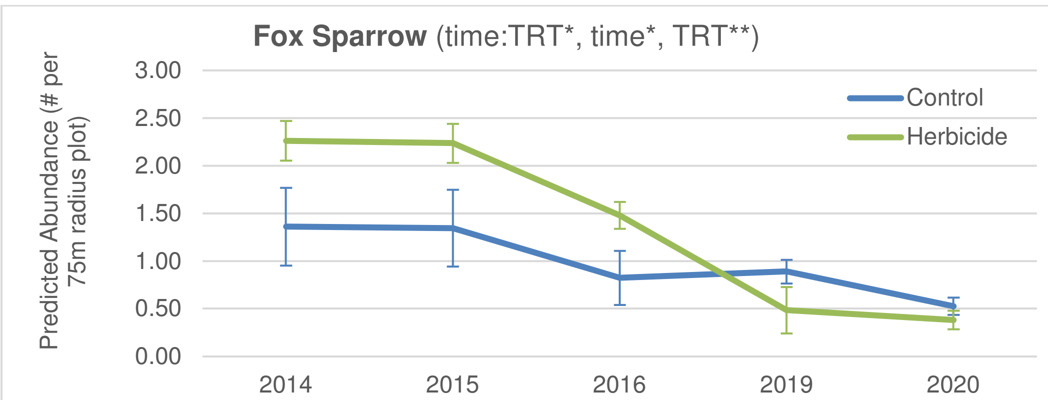
Figure 4. Summed CESF guild species abundance and richness. The "time" variable in ANOVA tests is year, and the treatment category (treatment sub-sample vs. control sub-sample) is abbreviated "TRT". ANOVA significance test indicated in chart titles as P<0.05 = *, P<0.005 = **, and P<0.005 = ***, non-significant = (ns). Treatments primarily occurred during 2018 with some follow-up herbicide spraying in 2019.

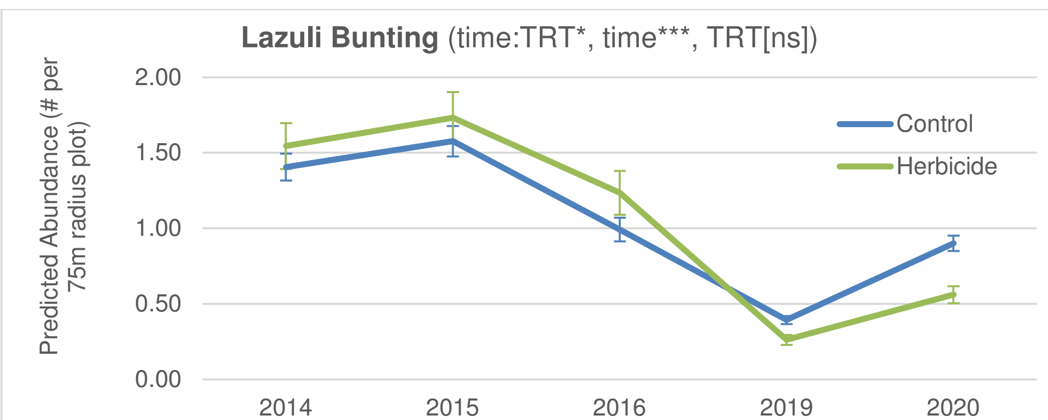


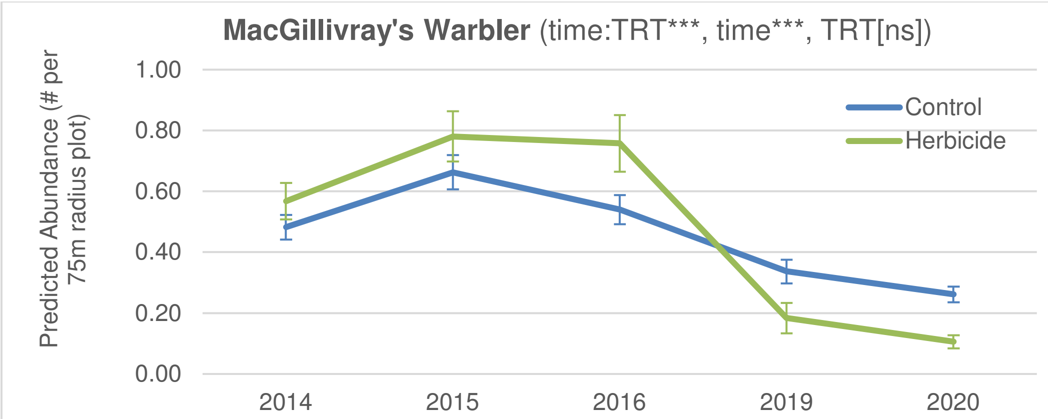


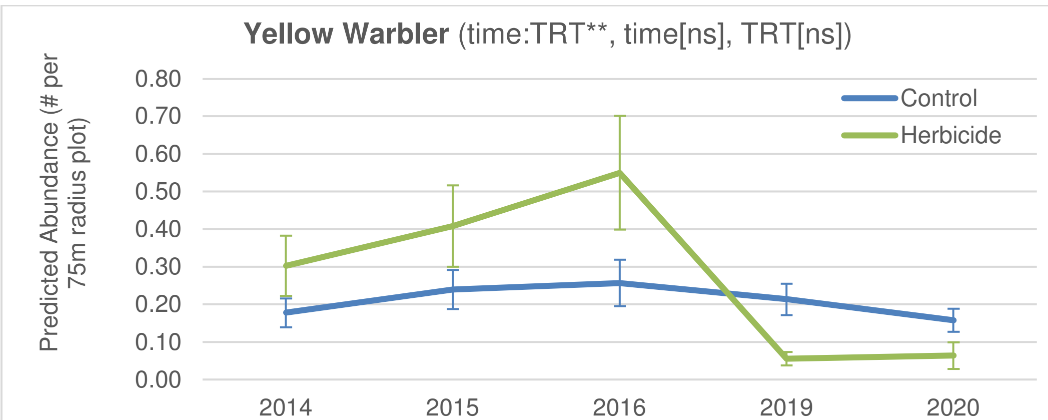
Five of the ten CESF species had significantly lower abundance following herbicide treatments (Fox Sparrow, Lazuli Bunting, MacGillivray's Warbler, Yellow Warbler and Mountain Quail) as indicated by a significant time*treatment interaction effect in ANOVA results (Figure 5). No species had significantly higher abundance in treated locations versus control locations. All but two species, Dusky Flycatcher and Mountain Quail, also showed an apparent declining trend over time, indicated by a significant time effect in ANOVA tests and a visual assessment of the abundance plots. No species showed an increasing trend, though Mountain Quail did have a significant time effect in the ANOVA test, this reflects variation across years and not an apparent trend. The treatment variable was significant for only two species, Fox Sparrow and Nashville Warbler, indicating that the control and treated samples may not have sampled equivalent habitat quality for those species leading to a difference in abundance.

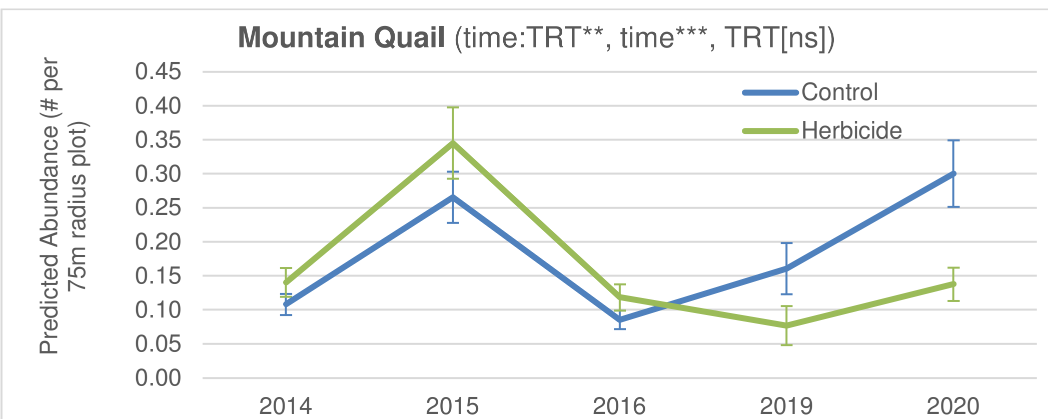
Figure 5. Summary of the time*treatment (TRT) effect for 10 CESF species. The "time" variable in ANOVA tests is years, and the treatment category (treatment sub-sample vs. control sub-sample) is abbreviated "TRT". The first five species (common name bolded) had a significant time:TRT effect indicating declines in herbicide-treated areas. The second five species (common name not bolded) did not have a significant effect. ANOVA significance test indicated in figure titles as P<0.05 = *, P<0.005 = **, and P<0.005 = ***, non-significant = (ns). Treatments primarily occurred during 2018 with some follow-up herbicide spraying in 2019.

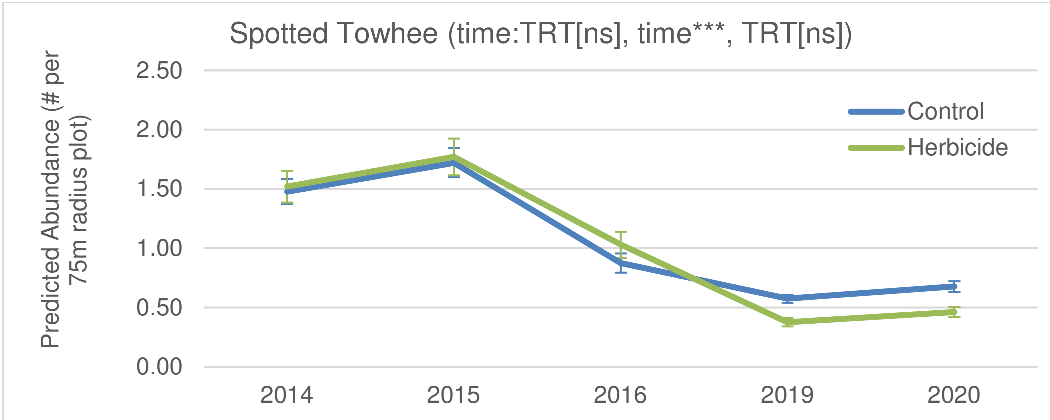


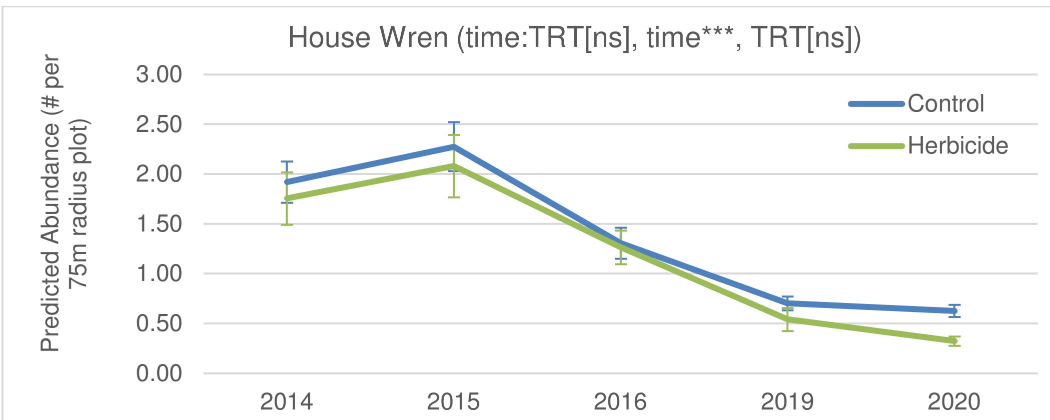


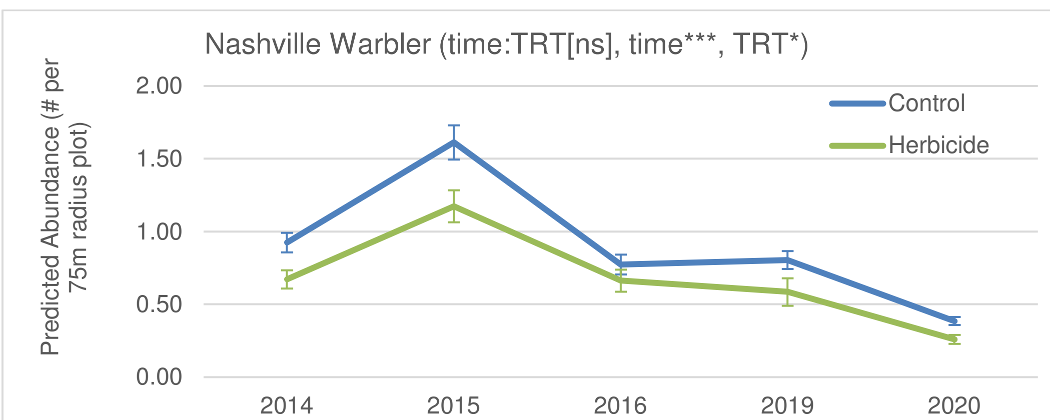

MacGillivray's Warbler (time:TRT***, time***, TRT[ns])
Predicted Abundance (# per 75m radius plot)
1.00
0.80
0.60
0.40
0.20
0.00
2014
2015
2016
2019
2020
Control
Herbicide

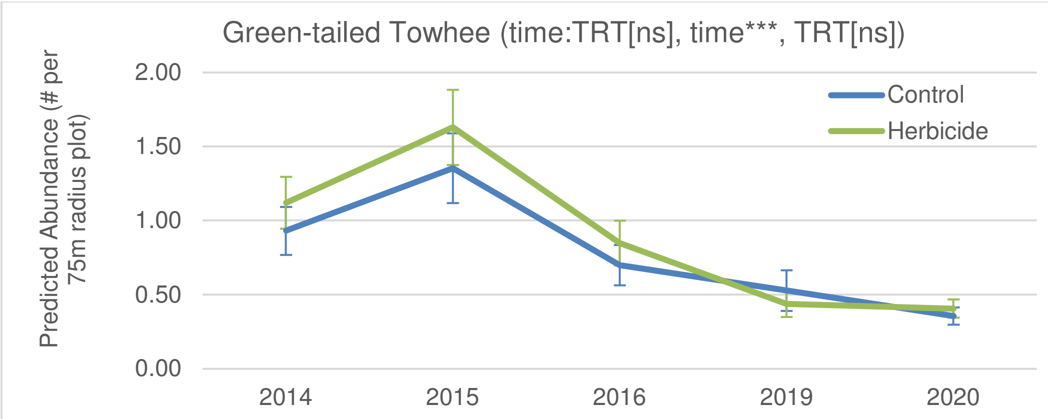

Yellow Warbler (time:TRT**, time[ns], TRT[ns])
Predicted Abundance (# per 75m radius plot)
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
2014
2015
2016
2019
2020
Control
Herbicide

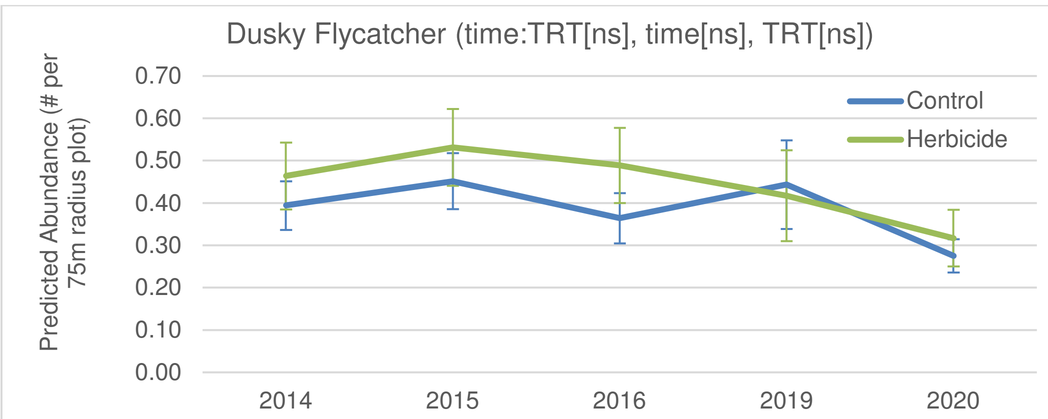

Mountain Quail (time:TRT**, time***, TRT[ns])
Predicted Abundance (# per 75m radius plot)
0.45
0.40
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
2014
2015
2016
2019
2020
Control
Herbicide


Spotted Towhee (time:TRT[ns], time***, TRT[ns])
Predicted Abundance (# per 75m radius plot)
2.50
2.00
1.50
1.00
0.50
0.00
2014
2015
2016
2019
2020
Control
Herbicide


House Wren (time:TRT[ns], time***, TRT[ns])
Predicted Abundance (# per 75m radius plot)
3.00
2.50
2.00
1.50
1.00
0.50
0.00
2014
2015
2016
2019
2020
Control
Herbicide


Nashville Warbler (time:TRT[ns], time***, TRT*)
Predicted Abundance (# per 75m radius plot)
2.00
1.50
1.00
0.50
0.00
2014
2015
2016
2019
2020
Control
Herbicide





DISCUSSION

Herbicide Effects

Herbicide treatments had a clear and large magnitude negative influence on CESF guild abundance and richness, as well as individually for most of the species we investigated. The effect of treatments was primarily driven by reductions in shrub cover, which was a significant predictor of abundance for many CESF species and was significantly reduced by the chainsaw and herbicide applications. While our study was not designed to identify immediate herbicide effects on wildlife, because we included shrub cover and a treatment factor in the abundance models as covariates, we can infer that the ancillary effects of herbicide treatment are much smaller influences on bird abundance than is the reduction in shrub cover. Other factors that may be driving the reduction in CESF abundance and richness in treated areas include toxicity of the chemicals to the birds themselves (including nestlings), and a wide variety of less-direct effects like increased toxicity of insect and plant forage, competition with other species for changed resource availability, presence of people and equipment during treatment activities, disturbances to the ground, soil, other plant materials, and other unmeasured processes. Food availability may be reduced since dead shrubs no longer produce seeds or host insect species that birds consume.

We also expected, and found, that strong yearly differences in bird abundance could be apparent in our data because of weather and climate-linked influences. Specifically, we found that the abundances of many species were measurably higher during the drought years of 2014-2016 (shown for example in: Roberts et al. 2019 and Saracco et al. 2019). Because of this potential confounding effect, we incorporated a time random effect into our abundance model and controlled for time in the ANOVA analysis in order to partition it from the treatment effect.

Other studies have shown deleterious effects from herbicide treatments on CESF bird community in western coniferous forests. CESF species like those in our study, including Lazuli Bunting and MacGillivray's Warbler, declined following herbicide treatments applied to young deciduous trees in conifer plantations in British Columbia (Easton and Martin 1998). Shrub nesting and foliage-gleaning bird species in SW Oregon declined in variable herbicide treatments in Douglas fir plantations, ranging from light, moderate and intensive reductions in early seral vegetation (Betts et al. 2013). Leaf gleaners also declined in intensive herbicide treatments in Douglas fir plantations in coastal Oregon up to 5 years post herbicide treatment (Kroll et al. 2017). Five of the 10 CESF species we analyzed are foliage gleaners that primarily forage within deciduous shrubs and all of these species require insect prey to provision their young. Presumably when these shrubs are killed, insect abundance declines as well (Sullivan and Sullivan 2003).

In other coniferous systems, bird species tended to persist in treated plantations if appropriate broadleaf plants were retained in 'green' islands (Santillo et al. 1989, MacKinnon and Freedman 1993), however abundance of understory species overall was greatly reduced in treated areas compared to untreated control units. In contrast to these studies, ours took place in regenerating conifer stands in burned forests where different successional processes may be occurring due to fire as the disturbance agent. We found few studies examining post-fire reforestation herbicide effects on the ecological community, although herbaceous plant diversity has shown to increase when herbicides are used to control shrub competition (Bohlman et al. 2016). The post-fire bird community, especially shrub nesters and foragers in moderate and high severity-burned areas, is a unique component in the Sierra Nevada landscape (Roberts et al. 2021). The Power Fire included approximately 5200 acres that burned at high severity on ENF land (with the remaining 1400 acres of high severity burned area on intensively managed private forest land). In the Power Fire Reforestation ROD (USDA Forest Service 2017), ENF proposed a 67% reduction in CESF habitat, a substantial proportion, especially considering there are no fires of similar age or size in the upper Mokelumne River watershed. Thus, these shrub-dependent bird species may not have had sufficient habitat to move to when the quality declined in herbicide-treated areas.

One important caveat is that our study took place only 1-2 years after herbicide application, so we note that delayed effects could be ecologically relevant. In the Freds Fire, we documented a weak negative herbicide effect where treatments took place 1-4 years before surveys (Fogg et al. 2016). Thus, we may see further reductions in CESF species in subsequent years after herbicide treatments, once seed bank densities decline and philopatric species (tendency to return to the same nesting locations year after year) have sufficient time to abandon no longer suitable habitat. The full effects of the herbicide treatments may take several more years to fully manifest. Migratory songbirds (which include most species in the CESF community) are generally philopatric; they return to nest the following year in the same area that they were born even if the habitat quality has changed (Greenwood 1980). Easton and Martin (1998) showed temporal lags in response to glyphosate herbicide treatments. As shrub skeletons are crushed by snow and break down and birds abandon unproductive territories, we might expect declines to continue for these species. In areas where the shrubs and hardwoods readily resprout and are not treated repeatedly with herbicides, bird species recovered within 2-3 years in coastal Oregon conifer forests (Kroll et al. 2017). However, the herbicides that were used to target the ceanothus species in Power Fire, and the season in which they were applied, are expected to permanently kill these species, lessening any chances of resprouting (Lanini and Radosevich 1982).

CESF Guild Habitat Associations

Most CESF species were more abundant in pre-treatment locations vs. control locations, indicating that the treatments were potentially targeting the highest quality CESF habitats (higher shrub cover relative to control locations). Most CESF species also had positive associations with live shrub cover, which was expected. Even so, the regression model results revealed much information on the mechanism of the effects of herbicide on CESF species. Of the 5 species with significantly lower abundance within the post-treatment sub-sample according to the ANOVA test, only Fox Sparrow had a significant negative effect of herbicide treatments (from the abundance regression model results). But four of the five (including also MacGillivray's Warbler, Yellow Warbler, and Mountain Quail) had strong positive coefficients for shrub cover indicating that the direct effect of herbicide on shrubs likely drives the lower abundance of these species in the post-treatment sub-sample. House Wren was unique in that the effect of herbicide treatment was significant and negative (from abundance regression model results), but they did not have significantly lower abundance in the post-treatment sample according to the ANOVA test. This likely results from non-significant coefficients for live and dead shrub cover and a positive effect for young conifer cover combining to maintain similar abundance within the treated sub-sample relative to the control locations.

Note that the coefficient for herbicide treatment was negative, though frequently non-significant, for all but one species (Green-tailed Towhee). This consistent pattern likely contributed to the significantly lower total abundance and richness in post-treatment locations for the CESF species. The effect of dead shrub cover was minimal for the CESF guild (with the exception of Fox Sparrow). Young conifer cover was also generally a smaller magnitude effect than live shrub cover for most species but was positive for both species for which it was significant (House Wren and Dusky Flycatcher). Given that most species responded negatively to the treatments, this suggests that increasing understory conifer cover does not adequately ameliorate (in terms of total guild abundance) the loss of broadleaf shrub cover. The one exception was Green-tailed Towhee for which understory conifer cover was significantly positive and for which there was no treatment effect. However, this species also had a strong positive association with shrub cover. Treated points with high young conifer cover may see a shift in guild composition towards more individuals of the species with positive association with that component, while species that are mostly shrub-dependent (e.g., Fox Sparrow, MacGillivray's and Yellow warblers) would be expected to decline leading to an overall decline in guild abundance. Finally, the treatment*salvage interaction model coefficient was significant for none of the CESF species. High standard errors for this coefficient across the guild, including one species with a non-fitting model when this variable was included, indicate that the effects of treatment within salvage logged locations is not different than unsalvaged locations.

Trends in CESF Bird Abundance

As mentioned in the introduction, the yearly changes in abundance are likely a result of weather and climate-linked effects where the abundances of many species were measurably higher during the drought years of 2014-2016 (Roberts et al. 2019, Saracco et al. 2019). Another confounding effect that could drive these yearly abundance changes is that several of these species reach peak abundance 13-15 years post-fire (e.g., House Wren, Spotted Towhee and Green-tailed Towhee; Taillie et al. 2018, Steel et al. 2021), thus some yearly changes in abundance across the 10 to 16-year post-fire period that we sampled within the Power Fire are to be expected. But we expect that the habitat conditions, given the amount of time post-fire for shrub development, should be nearly optimal for all the species within the CESF guild. Thus, the declines in CESF species in the herbicide-treated locations compared to controls within the Power fire during a period post-fire where we would expect species abundances to be stable if not increasing is notable. If there was a large amount of CESF within Power Fire, we may expect abundances at control locations to have a greater increase following treatments, but as treatments reduced approximately two-thirds of CESF habitat, there may not have been enough habitat available for birds to colonize. A more nuanced analysis of areas outside Power Fire and examining all bird species would help clarify what may be occurring with the yearly shifts in CESF bird abundance.

CONCLUSIONS

Herbicide treatments, which were largely targeted applications to deerbrush in chainsaw-released areas, in addition to areas of herbicide-only treatments dominated by whitethorn ceanothus and bear clover, do appear to have a strong negative effect on the CESF bird community, especially those dependent on shrub vegetation. Utilizing herbicides, especially within fire footprints where the majority of CESF habitat is targeted, can have deleterious effects on the CESF bird populations. If treatments are necessary, best practices include avoiding the breeding season (May-August), using a targeted approach with the smallest possible radius around young conifers, and leaving patches of intact shrubs. Evaluating the fire history of an entire watershed to help to ensure that CESF species have the potential to disperse to other areas of suitable habitat could inform decisions about the extent of CESF habitat within each fire footprint that would be appropriate for herbicide-based forest restoration treatments. When choosing areas at highest priority of reforestation and reducing shrub competition, consider whether these areas have the best chance of survival for conifers in a hotter, drier climate, especially on southwest slopes (North et al. 2019). Reintroducing fire to reforested areas can also be done in such a way to reduce young tree mortality (Bellows et al. 2016) but also reduce fuels (i.e., shrubs) that can lead to type conversion from forest to chapparal (Coppoletta et al. 2016). Lastly, several of the CESF species in the Sierra Nevada are declining according to Breeding Bird Survey long term trends from 1993-2019, including Fox Sparrow, Nashville Warbler, Yellow Warbler and Dusky Flycatcher (Sauer et al. 2020). Tradeoffs between growing trees, protecting those trees from fire damage, and reductions in biodiversity, especially declining species, should be balanced to promote ecological resilience of these fire-adapted forests and those species reliant on early successional habitat.

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