Prescribed Fire Limits Wildfire Severity Without Altering Ecological Importance for Birds

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Research Article

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Abstract

Background: Fire suppression and anthropogenic land use have increased severity of wildfire in dry conifer forests. Managers use fuels reduction (e.g., prescribed fire) to limit high-severity wildfire and restore ecological function to fire-adapted dry conifer forests. Many species that evolved in these forests, however, are adapted to high-severity wildfire. To fully understand the ecological implications of fuels reduction treatments, we need to understand direct treatment effects and how treatments modulate subsequent wildfire effects on natural communities. We studied bird population and community patterns over 9 years, including unburned (2002–2003), after prescribed fire (2004–2007), and after wildfire (2008–2010). We used a before-after, control-impact (BACI) approach to analyze shifts in species occupancy and richness with respect to treated units following prescribed fire and burn severity following wildfire.

Results: We found both positive and negative wildfire and prescribed fire effects on birds largely consistent with species life history traits; several woodpecker species, secondary cavity-nesting species, aerial insectivores, and understory species exhibited positive effects, whereas open cup canopy-nesting species and foliage- or bark-gleaning insectivores exhibited negative effects. Also as expected, wildfire affected more species more consistently than prescribed fire. Wildfire burned units initially treated with prescribed fire less severely than untreated units, but wildfire effects on birds were similar regardless of treatment.

Conclusions: Our results suggest managers can employ prescribed fire to reduce wildfire severity without necessarily altering the ecological importance of wildfire to birds. Additional study of the ecological implications of various fuels reduction practices, representing a range of intensities and fire regimes, would further inform forest management that includes biodiversity objectives.

Background

Wildfire strongly shapes the amount and distribution of biodiversity in western North American forests. Some species occur more frequently and others less frequently in recently burned forest, causing community composition to vary with burn severity (Saab et al. 2005; Kalies et al. 2010; Fontaine and Kennedy 2012). Landscapes containing a diversity of stands varying in history of fire are therefore expected to support the greatest array of species (Clarke 2008; Fontaine et al. 2009; Fontaine and Kennedy 2012). Within the last ~100 years, anthropogenic fire suppression, logging, development, livestock grazing, and climate change have caused fuel accumulation and homogenization of vegetation structure in many lower elevation dry conifer forests (Covington and Moore 1992; Agee 1993; Brown et al. 2004; Schoennagel et al. 2004). These changes have shifted fire regimes towards less frequent but larger and more severe wildfire, with potential negative consequences for the economic and aesthetic values of forests, human safety, and wildlife diversity (Dale et al. 2001; Brown et al. 2004; Schoennagel et al. 2004).

Forest managers widely implement fuels reduction treatments, i.e., prescribed fire, timber harvest, or some combination of both, to limit wildfire size and severity, with the ultimate goal of restoring historical forest conditions and mitigating anthropogenic impacts (Fulé et al. 2012). Empirical studies confirm expected reductions in wildfire severity in treated areas for a limited number of years following treatment, particularly when fuel loads are greatly reduced (Pollet and Omí 2002; Fulé et al. 2012; Prichard and Kennedy 2014; Fernandes 2015). Thus, strategically placed treatments could help managers reduce the extent of subsequent wildfire (Stevens et al. 2014). Some expect this approach to restore historical conditions and ecological function to many dry conifer forests (Walker et al. 2018). Desirable historical conditions are difficult to achieve because of climate-induced ecological changes (McKelvey et al. 2021), and because they vary regionally and by spatial scale (Schoennagel et al. 2004; Bock and Block 2005; Illán et al. 2014). Animal ecologists therefore suggest treatments could be ineffective or inappropriate in regions like the central Rocky Mountains where historical levels of diversity were associated with relatively heterogeneous landscapes maintained by mixed severity fire (Saab et al. 2005; Latif et al. 2016b).

Birds are valuable as focal organisms for understanding faunal community relationships with wildfire and forest management. Surveys do not require specialized equipment (Sutherland et al. 2004), allowing birds to readily inform management strategies aimed at biological conservation (Saab and Powell 2005; Saab et al. 2005). Additionally, hierarchical occupancy models facilitate analysis of survey data to evaluate population and community relationships with environmental disturbance and management treatments (Dorazio et al. 2006; Russell et al. 2009; Russell et al. 2015; Latif et al. 2016b). Bird responses to disturbance depend on species ecology and life history traits (Saab and Powell 2005; Smucker et al. 2005; Kotliar et al. 2007; Fontaine and Kennedy 2012; Seavy and Alexander 2014). Wildfire opens the canopy, which can stimulate understory vegetative growth and improve foraging and nesting opportunities for shrub-nesting and ground-foraging species, and wildfire creates snags that provide important nesting and foraging resources for cavity-nesting species (Hutto 1995; Kotliar et al. 2002; Saab et al. 2009). In contrast, tree mortality after wildfire reduces resources for canopy-nesting species and species that forage on live trees (Kotliar et al. 2007; Fontaine et al. 2009). As with wildfire communities in general, ecologists expect landscapes representing the historical range of fire conditions to support the greatest array of bird species (see reviews by Kalies et al. 2010; Fontaine and Kennedy 2012).

Fuels reduction treatments could shape bird diversity in various ways. Birds may respond directly to treatment-induced changes in forest structure (Russell et al. 2009; Gaines et al. 2010; Fontaine and Kennedy 2012; White et al. 2013). Managers sometimes look to fuels treatments to provide a surrogate for wildfire (McIver et al. 2013), but severity and scale limit the potential for treatment effects to emulate wildfire effects on birds (Fontaine and Kennedy 2012). Instead, current debate focuses more on how treatments modulate burn severity and consequent wildfire effects on birds (Hutto et al. 2014). By altering subsequent wildfire behavior, fuels reduction treatments can change the ecological significance of wildfire for birds. For example, by limiting high-severity crown fire, wildfire in treated areas may not generate enough snags to benefit cavity-nesting species or open the canopy sufficiently to benefit understory species (Hutto et al. 2015). Conversely, reduced tree mortality may result in limited negative impacts of wildfire for canopy-nesting and foliage-gleaning species. Studies comparing wildfire behavior and bird responses to wildfire in treated versus untreated stands are needed to test these hypotheses. Knowledge of how fuels reduction treatments directly and indirectly influence avian populations and communities will inform forest management activities that incorporate habitat conservation for avian diversity.

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We studied avian relationships with prescribed fire and wildfire in the Payette National Forest (NF), a lower elevation dry conifer forest in the central Rocky Mountains historically associated with a mixed severity fire regime. We surveyed birds in paired treatment and control study units before (2002–2003) and after (2004–2007) prescribed fire, and following wildfire (2008–2010). We evaluated two primary hypotheses: 1) wildfire burn severity would be lower in units initially treated with prescribed fire, and consequently 2) birds would respond differently to wildfire in treated compared to untreated units. Secondarily, being more severe and extensive, we expected wildfire to have stronger effects on bird populations than prescribed fire. We also built on published literature and evaluated hypotheses therein regarding expected responses to wildfire and prescribed fire for particular life histories (Russel et al. 2009; Latif et al. 2016b). We primarily evaluated our hypotheses by looking for temporal shifts in species occupancy of sites varying in burn severity following disturbance, providing relatively strong inference of disturbance effects (Popescu et al. 2012; Russell et al. 2015). We considered implications of observed patterns for forest management with objectives that include conservation of avian diversity.

Methods
Study system
The Payette NF is in the central Rocky Mountains of western North America (45°00′30″ N 116°02′30″ W; elevation 1127 m – 2075 m). The East Zone Complex Fire burned the Payette NF in July–October, 2007 (95,100 ha; Fig. 1). About 60 years prior to this study, forest managers began suppressing wildfire and managing for multiple uses, including timber harvest, mining, recreation, livestock grazing, wildlife habitat, and watershed management (Hollenbeck et al. 2013). Following the classification scheme of Miller and Thode (2007), burn severity within the East Zone Complex Fire perimeter was classified as 9% unburned, 19% low severity, 26% moderate severity, and 46% high severity (see also Latif et al. 2016b).

The canopy was dominated by large (≥ 23 cm dbh) ponderosa pine (Pinus ponderosa) trees (> 65%; Hollenbeck et al. 2013). Other tree species included Douglas-fir (Pseudotsuga menziesii), lodgepole pine (Pinus contorta) and small patches (< 10 ha) of quaking aspen (Populus tremuloides) in snowmelt drainages. Common understory species include snowberry (Symphoricarpos albus), spirea (Spirea betulifolia), Saskatoon serviceberry (Amelanchier alnifolia), and chokecherry (Prunus virginiana).

Study units
We established six study units distributed across ~ 20,000 ha (Fig. 1). Study units were delineated in pairs so that members of each pair were similar in vegetation and topography, and 1 member of each pair was randomly selected for prescribed fire treatment (Table 1) (for additional details, see Saab et al. 2007). Forest managers applied prescribed fire treatments in spring prior to the breeding season for most bird species (April – early May); one unit was treated in 2004, and the other two in 2006. The East Zone Complex Fire subsequently burned 5 units (2 treatment, 3 control), where we collected post-wildfire data for another 3 years.

| Pair | Study unit | n points | Years surveyed | Prescribed fire timing | Wildfire timing |
|------|------------|----------|----------------|------------------------|----------------|
| 1    | Buckhorn   | 20       | 2002–2010      | Control                | Fall, 2007     |
|      | Fitsum Creek | 20       | 2002–2010      | Spring, 2006          | Fall, 2007     |
| 2    | Williams   | 10       | 2002–2010      | Control                | Fall, 2007     |
|      | Dutch Oven | 20       | 2002–2010      | Spring, 2006          | Fall, 2007     |
| 3    | Deadman    | 20       | 2002–2006, 2008–2010 | Control            | Fall, 2007     |
|      | Parks Creek | 20       | 2002–2006      | Spring, 2004          | not surveyed   |

Bird surveys
We surveyed birds at 110 point survey stations distributed across the 6 study units (Table 1, Fig. 1). We spaced survey points at least 150 m apart (mean = 277, SD = 68) within study unit boundaries. For statistical independence, we space most points ≥ 200 m apart, but we were forced to space a minority of points (32%) in closer proximity due to steep topography with limited access for humans. We surveyed each point twice each between 23 May and 3 July over 2 years (2003–2004), another 3 years.

We surveyed birds through 2006 and the remaining two pairs through 2007, yielding 2–4 years pre- and 2–3 years post-treatment data with respect to prescribed fire (Table 1). We then continued surveys in the 5 units burned by wildfire for an additional 3 years.

Burn severity measurements and analysis
We measured burn severity using the composite burn severity index (CBI; Key and Benson 2006) modified to accommodate our study area and objectives. We calculated a CBI value for each survey point (0–3 range) representing the mean of up to 11 components quantifying aspects of canopy structure, understory vegetation and topography, and 1 member of each pair was randomly selected for prescribed fire treatment (Table 1) (for additional details, see Saab et al. 2007). Forest managers applied prescribed fire treatments in spring prior to the breeding season for most bird species (April – early May); one unit was treated in 2004, and the other two in 2006. The East Zone Complex Fire subsequently burned 5 units (2 treatment, 3 control), where we collected post-wildfire data for another 3 years.
after prescribed fire (2004–2007), and after wildfire (2008–2010). Components represented either changes in these attributes from before to after disturbance, or aspects of burn severity apparent after disturbance (e.g., extent of char). We only measured points within burned units and assumed CBI = 0 for units that were not burned during our study. Others describe in detail how CBI values correspond with changes in various aspects of vegetation structure (Key and Benson 2006; Saab et al. 2006). In short, CBI = 0, 0 < CBI < 1, 1 < CBI < 2.25, and CBI > 2.25 are interpretable as unburned, low severity, moderate severity, and high severity, respectively. In general, low severity fire primarily affects understory vegetation with minimal canopy mortality (<40%), whereas high severity fire results in much greater canopy mortality (>70%).

We quantified prescribed fire CBI (hereafter CBI_{PF}) using environmental data collected before (2002–2003) versus after (2004–2007) prescribed fire, and wildfire CBI (hereafter CBI_{WF}) with data from immediately before (2004–2007) versus after wildfire (2008–2010). Unfortunately, wildfire burned one treated study unit (Fitsum Creek) before we could measure post-treatment prescribed fire conditions (2004–2007). For this unit, we imputed CBI_{WF} by 1) calculating CBI_{total} representing overall burn severity (i.e., changes from 2002–2003 to 2008–2010), 2) regressing CBI_{WF} as a linear function of CBI_{total} at units where both were available (Buckhorn, Dutch Oven, Williams, and Deadman), and 3) using the resulting regression model (CBI_{WF} = β_0 + β_1 × CBI_{total} with estimates [β_0 [s.e.] = -0.04623 [0.07653] and β_1 = 0.98415 [0.04271]) to impute missing data. As a covariate of occupancy, we imputed missing CBI_{WF} values using a normally distributed prior with mean and SD representing model-predicted CBI_{WF}.

Data for calculating CBI_{PF} were relatively limited, so we did not use CBI_{PF} as a covariate of occupancy. Rather, we modeled occupancy with a categorical treatment effect (Trt_{PF} = 0 or 1 for survey points in untreated versus treated units, respectively). We then summarized CBI_{PF} values where available (Dutch Oven, Parks Creek) to inform inference and compare with CBI_{WF}. We compared CBI_{WF} between treated and untreated points within treatment-control unit pairs and related CBI_{WF} with CBI_{PF} where available to evaluate the effect of prescribed fire treatments on subsequent wildfire severity.

### Occupancy models

We analyzed avian relationships with prescribed fire using community occupancy models formulated within a Bayesian hierarchical modeling framework (Dorazio et al. 2006; Russell et al. 2009). Occupancy models leverage repeat-survey data to estimate species detectability (ψ) conditional upon occupancy (species occurrence within a specified time period), allowing unbiased estimation of occupancy probabilities (ψ) given sufficient data and adherence to model assumptions (MacKenzie et al. 2002; MacKenzie et al. 2006). We assumed that the occupancy states of species could change among years, but not between visits within a year. We estimated species-specific parameters as random variables governed by community-level parameters. The use of a common distribution among species improves the precision of species-specific parameter estimates, particularly for rare species (Dorazio et al. 2006, Russell et al. 2009). We excluded raptors, owls, and grouse from analysis because they were not readily detectable with our survey methods. Additionally, we only included species that bred in our study areas. For mobile animals such as birds, detectability (ψ) estimated with surveys repeated over a season quantifies both within-season movement and the observation process (i.e., availability and perceptibility; sensu Chandler and Royle 2013; Amundson et al. 2014). In principle, occupancy probabilities thereby estimated the probability of a surveyed point intersecting ≥ 1 home range for a given species (Efford and Dawson 2012; Latif et al. 2016a).

We compiled a 3-dimensional data matrix y, where element y_{ijt} was the sum of binary indicators for species detection (Sanderlin et al. 2014). Given a binary indicator x_{ijt} = 1, we detected species i (i = 1,…,N) at survey point j (j = 1,…,J) during visit k (k = 1,…,K) in year t (t = 1,…,T, T = 4). Because we did not have covariates that differed for detection between visits, we analyzed the sum of all binary detections for species i over all visits at each survey point j in year t, where y_{ijt} = ∑_k x_{ijkt} and y_{ijt} ∈ [0,1,…,K]. We modeled these data given probability of detection ρ_i and occupancy latent state z_i at a Bernoulli distribution with probability of success p_i × z_i |

\[ y_{ijt} | p_i, z_i ] \sim Bernoulli(p_i z_i) \] (Equation 1),

where the latent variable z_i, for occupancy given probability of occupancy ψ_i, was modeled as:

\[ z_i | ψ_i ] \sim Bernoulli(ψ_i) \] (Equation 2).

We analyzed changes in species occupancy patterns using a model that leveraged our before-after, control-impact (BACI) sampling for examining disturbance effects (Popescu et al. 2012; Russell et al. 2015). For prescribed fire effects, we modeled occupancy (ψ_{ijt}) as a function of treatment (Trt_{PF}), period (Per_{P} = 0 or 1 for before or after treatment, respectively), and the interaction of the two (Trt_{PF} × Per_{P}). Thus,

\[ \logit(ψ_{ijt}) = β_{0,i} + β_{PerPF,i} \times Per_{PF,t} + β_{TrtPF,i} \times Trt_{PF,j} + β_{PerPF \times TrtPF,i} \times Per_{PF,t} \times Trt_{PF,j} \] (Eq. 3),

where β_{0,i} is the intercept and β_{PerPF,i}, β_{TrtPF,i} and β_{PerPF \times TrtPF,i} describe the additive and interactive effects of covariates Per_{PF,i} and Trt_{PF,j} on occupancy of species i at survey point j in year t. We restricted analysis of prescribed fire effects to data collected before wildfire (2002–2007). For wildfire effects, we analyzed data collected 2 years before and 3 years after wildfire (2006–2010) using two models. The first model analyzed overall wildfire effects:

\[ \logit(ψ_{ijt}) = β_{0,i} + β_{PerWF,i} \times Per_{WF,t} + β_{CBIWF,i} \times CBI_{WF,j} + β_{PerWF \times CBIWF,i} \times Per_{WF,t} \times CBI_{WF,j} \] (Eq. 4).
\[
\logit \left( \psi_{ij} \right) = \beta_{0,ij} + \beta_{\text{Per}_{pp},i} \times \text{Per}_{WF},t + \beta_{\text{Trt}_{pp},i} \times \text{Trt}_{PF},t + \beta_{\text{CBI}_{WF},i} \times \text{CBI}_{WF},j + \beta_{\text{Per}_{pp} \times \text{CBI}_{WF},i} \times \text{Per}_{WF},t + \beta_{\text{Trt}_{pp} \times \text{CBI}_{WF},i} \times T_i.
\] (Eq. 5)

As in Eq. 3, \( \beta_{0,ij} \) is the intercept and all remaining \( \beta \) parameters describe additive and interactive effects of covariates on avian occupancy in Equations 4 and 5. All estimated parameters were species-specific normal random effects. For numerical purposes, we rescaled CBI\(_{WF}\) values to mean = 0 and SD = 1 prior to analysis.

For all three models above (Equations 3–5), we drew inference of disturbance (prescribed fire or wildfire) effects from the extent to which occupancy shifted towards or away from burned (or unburned) survey points following disturbance. Interaction parameters in Equations 3, 4, and 5 quantified these shifts, whereas additive parameters controlled for potentially confounding environmental variation among survey points and time periods (Popescu et al. 2012). We considered statistically supported interaction parameters (90% Bayesian credible interval [BCI] excluded zero) strong evidence for disturbance effects.

We used one additional model to analyze annual changes in bird occupancy and time-dependent disturbance effects with all available data (2002–2010). This model included a random year effect and year-specific prescribed fire and wildfire effects:

\[
\logit \left( \psi_{ij} \right) = \beta_{0,ij} + \beta_{\text{Trt}_{pp},t} \times \text{Trt}_{PF},j + \beta_{\text{CBI}_{WF},t} \times \text{CBI}_{WF},j
\] (Eq. 6)

The intercept, \( \beta_{0,ij} \), varied with species and year according to nested normal random effects (year within species). Prescribed fire effects (\( \beta_{\text{Trt}_{pp},t} \)) were estimated separately for 4 distinct time periods, pre-treatment (\( \text{tp}_p = 0 \) and 1–3 years post-treatment (\( \text{tp}_p = 1–3 \), respectively). Similarly, wildfire effects (\( \beta_{\text{CBI}_{WF},t} \)) were estimated for 4 time periods, fire-pre (2006–2007, \( \text{tf}_w = 0 \)) and 1–3 years post-fire (2008–2010, \( \text{tf}_w = 1–3 \), respectively). \( \beta_{\text{Trt}_{pp},t} \) was not estimated for 2008–2010 and \( \beta_{\text{CBI}_{WF},t} \) was not estimated for 2002–2005 for comparability with other models (see Equations 3–5). We used this model to look for time-dependencies in disturbance effects (i.e., where 95% BCI for \( \beta_{\text{Trt}_{pp},t} \) \(( \text{tp}_p \in (1, 2, 3) \)) or \( \beta_{\text{CBI}_{WF},t} \) \(( \text{tf}_w \in (1, 2, 3) \)) excluded zero, Eq. 6). Additionally, we scanned yearly occupancy estimates for surveyed sites (\( \psi^\prime_t = \sum_{j=1}^{J} z_{ij,t} / J \)) to identify notable changes among pre-treatment (2002–2003), post-treatment (2004–2007), and post-wildfire (2008–2010) periods. All surveyed sites were burned by wildfire to some degree (min CBI = 0.39, see Results), so we expected some changes in overall occupancy for species with similar responses to low- versus high-severity wildfire. We considered inference from changes in overall occupancy weaker, however, because estimates of these changes did not control for potentially confounding factors as did shifts in occupancy with respect to CBI (see above).

In addition to species-specific relationships, we plotted emergent changes between species richness with treatment condition. We estimated species richness (\( N_j \)) at each survey point \( j \) and year \( t \) \( N_{jt} = \sum_{i=1}^{\text{max}(i)} z_{ijt} \). Similar to some (Russell et al. 2009, Latif et al. 2016) and unlike others (Dorazio et al. 2006, Kéry et al. 2009), we did not augment data to represent unsampled species; so community-level inferences were restricted to the subset of species observed at least once during our studies.

We modeled detectability as a species-specific normal random effect \( b_{0,i} \):

\[
\logit(p_i) = b_{0,i}, \quad (\text{Eq. 7}),
\]

where \( p_i \) is the probability of detecting species \( i \) when surveying a given survey point in a given year when the species was present (i.e., \( \geq 1 \) home range intersected the 100m point neighborhood). Unlike others (Russell et al. 2015), we did not consider treatment effects on detectability. Estimated effects on detectability from preliminary analyses were imprecise (credible intervals overlapped 0 for all species) and model convergence was difficult to achieve, suggesting the additional complexity straining limits of the data (Q. Latif unpublished data). We therefore only modeled heterogeneity in detectability among species and assumed detectability did not change with treatment condition. We modeled heterogeneity among species using a correlation term (\( \rho \)) between species intercepts of detection probability (\( b_{0,i} \)) with occupancy probability (\( \beta_{0,i} \)) (Dorazio and Royle 2005, Kéry et al. 2009).

We sampled posterior parameter distributions for all models using JAGS v4 (Plummer 2003) programmed from R (R Core Team 2013, Su and Yajima 2014). We used independent non-informative priors for all parameters (for priors, see Appendix B). For each model, we ran 6 parallel MCMC chains of length 100,000 \( it \) burn-in 10,000 \( it \) and thinning 100 \( it \) to sample posterior distributions. We verified that \( n_{\text{effective}} \geq 100 \) and \( R \leq 1.1 \) for all parameters (Gelman and Hill 2007). We examined model goodness-of-fit (GOF) using posterior predictive testing (Gelman and Hill 2007). Specifically, we calculated a Bayesian \( p \)-value representing the proportion of simulated datasets drawn from model posterior predictive distributions with deviance higher than deviance for observed datasets from each location, where \( p < 0.05 \) or \( p > 0.95 \) constituted evidence for lack of fit.

**Results**

We detected 60 species across all survey points and years (Table 2). The five most frequently detected species were Yellow-rumped Warbler, Western Tanager, Chipping Sparrow, Hammond’s Flycatcher, and Red-breasted Nuthatch. The distributions of CBIs for wildfire and prescribed fire at surveyed points broadly overlapped, but on average wildfire was more severe especially in areas not initially treated with prescribed fire (Fig. 2). Severe wildfire (CBI > 2) was extensive at untreated units (10 of 30 points) in contrast with wildfire and prescribed fire at treated units (1 of 40 points for each; Fig. 2). Species-specific detection probability estimates varied and correlated moderately with occupancy (Appendix C). We found no evidence for lack of fit for community occupancy models.
Table 2

Number of detections (No. point × year occasions detected) recorded for species observed 2002–2010 in the Payette National Forest, Idaho. The maximum number of detections possible for each fire condition ( unburned, treated units after prescribed fire, and after wildfire) are noted in header parentheses.

| Common name                  | Taxonomic name      | Code  | No. point × year occasions detected (max possible) |
|------------------------------|---------------------|-------|--------------------------------------------------|
|                              |                     |       | unburned (360)       | prescribed fire (260) | wildfire (270) |
| Mourning Dove                | Zenaida macroura    | MODO  | 0       | 3       | 8       |
| Common Nighthawk             | Chordeiles minor    | CONI  | 1       | 3       | 2       |
| Broad-tailed Hummingbird     | Selasphorus platycercus | BTHU  | 2       | 1       | 3       |
| Rufous Hummingbird           | Selasphorus rufus   | RUHU  | 1       | 0       | 1       |
| Calliope Hummingbird         | Selasphorus calliope| CAHU  | 18      | 4       | 1       |
| Williamson's Sapsucker       | Sphyrapicus thyroideus | WISA | 6       | 0       | 1       |
| Red-naped Sapsucker          | Sphyrapicus nuchalis| RNOSA | 3       | 1       | 3       |
| Downy Woodpecker             | Dryobates pubescens | DOWO  | 0       | 0       | 1       |
| Hairy Woodpecker             | Dryobates villosus  | HAWO  | 64      | 35      | 75      |
| American Three-toed Woodpecker | Picoides dorsalis | ATTW  | 0       | 5       | 1       |
| Black-backed Woodpecker      | Picoides arcticus   | BBWO  | 2       | 5       | 6       |
| Northern Flicker             | Colaptes auratus    | NOFL  | 33      | 17      | 25      |
| Pileated Woodpecker          | Dryocopus pileatus  | PIWO  | 4       | 4       | 10      |
| Olive-sided Flycatcher       | Contopus cooperi    | OSFL  | 3       | 7       | 5       |
| Western Wood-Pewee           | Contopus sordidulus | WEWP  | 8       | 2       | 7       |
| Hammond's Flycatcher         | Empidonax hammondii | HAFL  | 275     | 178     | 155     |
| Dusky Flycatcher             | Empidonax oberholseri | DUFL | 55      | 37      | 64      |
| Cassin's Vireo               | Vireo cassini       | CAVI  | 90      | 42      | 44      |
| Warbling Vireo               | Vireo gilvus        | WAVI  | 72      | 71      | 62      |
| Canada Jay                   | Perisoreus canadensis | CAJA | 14      | 6       | 3       |
| Steller's Jay                | Cyanocitta stelleri | STJA  | 10      | 9       | 12      |
| Clark's Nutcracker           | Nucifraga columbiana| CLNU  | 21      | 19      | 16      |
| Common Raven                 | Corvus corax        | CORA  | 5       | 3       | 2       |
| Black-capped Chickadee       | Poecile atricapillus| BCCH  | 4       | 0       | 0       |
| Mountain Chickadee           | Poecile gambeli     | MOCH  | 165     | 106     | 107     |
| Red-breasted Nuthatch        | Sitta canadensis    | RBNU  | 235     | 175     | 172     |
| White-breasted Nuthatch      | Sitta carolinensis  | WBNU  | 25      | 11      | 20      |
| Brown Creeper                | Certhia americana   | BCR  | 50      | 15      | 9       |
| Rock Wren                    | Salpinctes obsoletus| ROWR  | 5       | 6       | 16      |
| Canyon Wren                  | Catharpes mexicanus | CANW  | 2       | 0       | 0       |
| House Wren                   | Troglydites aedon   | HOWR  | 2       | 0       | 14      |
| Pacific Wren                 | Troglydites pacificus | PAWR | 10      | 4       | 0       |
| Golden-crowned Kinglet       | Regulus satrapa     | GCKI  | 2       | 2       | 0       |
| Ruby-crowned Kinglet         | Regulus calendula   | RCKI  | 25      | 13      | 5       |
| Western Bluebird             | Sialia mexicana     | WUBL  | 0       | 1       | 2       |
| Mountain Bluebird            | Sialia currucoides  | MOBL  | 3       | 1       | 36      |
| Townsend’s Solitaire         | Myaestes townsendi  | TOSO  | 73      | 47      | 29      |
| Swainson’s Thrush            | Catharus ustulatus  | SWTH  | 49      | 46      | 25      |
| Hermit Thrush                | Catharus guttatus   | HETH  | 47      | 61      | 48      |
breasted Nuthatch occupancy was distributed in completely opposite directions with respect to wildfire burn severity in years 1 (negative) versus 3 (positive). Of particular note, White-breasted Nuthatch, Hammond’s Flycatcher, Lazuli Bunting, and Pine Siskin initially shifted towards lower severity burned points in year 1 but then shifted back in years 2–3 after wildfire. White-breasted Nuthatch, Hammond’s Flycatcher, Lazuli Bunting, and Pine Siskin exhibited lagged shifts towards untreated units in years 2 or 3 following prescribed re. We found statistically supported prescribed re effects for 2 species and wildfire effects for 7 species (Fig. 6). These prescribed re effects translated into smaller changes in occupancy than did wildfire effects (compare Fig. 6 with Figs. 4, 5).

Wildfire effects on species occupancy were similar in units previously treated with prescribed re compared to untreated units (Fig. 7). The data did not definitively support differences in wildfire effect between treated and untreated units for any species (all 95% BCIs for $\beta_{\text{PERW}}$, $\beta_{\text{TWWF}}$, and $\beta_{\text{CBIWF}}$ from Eq. 5 included zero). Additionally, species exhibiting the strongest shifts towards or away from high-severity burned points after wildfire were the same in units initially treated versus untreated with prescribed re (Fig. 7). For certain species, statistical support for wildfire effects differed with prior treatment (see Mountain Chickadee, Cassin’s Vireo, and Warbling Vireo). Nevertheless, species never exhibited completely contradictory effects in treated versus untreated units.

Overall prescribed fire and wildfire effects

We found statistically supported prescribed fire effects for 2 species and wildfire effects for 7 species (Fig. 3). House Wren, Hairy Woodpecker, Olive-sided Flycatcher, and Brewer’s Sparrow occupancy shifted towards high-severity burned points following wildfire (Fig. 4). Conversely, Cassin’s Vireo, Townsend’s Warbler, and Warbling Vireo occupancy shifted towards lower severity burned points (Fig. 5). Rock Wren and American Three-toed Woodpecker shifted towards treated units after prescribed fire (Fig. 6). These prescribed fire effects translated into smaller changes in occupancy than did wildfire effects (compare Fig. 6 with Figs. 4, 5).

Overall prescribed fire and wildfire effects

| Common name | Taxonomic name | Code | No. point × year occassions detected (max possible) |
|-------------|----------------|------|--------------------------------------------------|
|             |                |      | unburned (360) | prescribed fire (260) | wildfire (270) |
| American Robin | Turdus migratorius | AMRO | 171 | 104 | 105 |
| Cedar Waxwing | Bombycilla cedrorum | CEDW | 1 | 2 | 0 |
| Orange-crowned Warbler | Leiothlypis celata | OCWA | 8 | 8 | 5 |
| Nashville Warbler | Leiothlypis ruficapilla | NAWA | 34 | 28 | 12 |
| MacGillivray’s Warbler | Geothlypis tolmiei | MGWA | 67 | 49 | 72 |
| Yellow Warbler | Setophaga petechia | YEWA | 0 | 1 | 1 |
| Yellow-rumped Warbler | Setophaga coronata | YRWA | 313 | 213 | 209 |
| Townsend’s Warbler | Setophaga townsendi | TOWA | 64 | 35 | 37 |
| Spotted Towhee | Pipilo maculatus | SPTO | 4 | 3 | 1 |
| Chipping Sparrow | Spizella passerina | CHSP | 281 | 168 | 233 |
| Brewer’s Sparrow | Spizella breweri | BRSP | 0 | 7 | 3 |
| Song Sparrow | Melospiza melodia | SOSP | 4 | 4 | 6 |
| Lincoln’s Sparrow | Melospiza lincolni | LISP | 0 | 2 | 6 |
| Dark-eyed Junco | Junco hyemalis | DEJU | 152 | 74 | 127 |
| Western Tanager | Piranga ludoviciana | WETA | 306 | 215 | 213 |
| Black-headed Grosbeak | Pheucticus melanocephalus | BHGR | 8 | 3 | 22 |
| Lazuli Bunting | Passerina amoena | LAZB | 3 | 1 | 74 |
| Cassin’s Finch | Haemorhous cassinii | CAFI | 98 | 89 | 118 |
| Red Crossbill | Loxia curvirostra | RECR | 68 | 25 | 67 |
| Pine Siskin | Spinus pinus | PISI | 53 | 14 | 112 |
| Evening Grosbeak | Coccothraustes vespertinus | EVGR | 3 | 2 | 1 |

Annual changes in occupancy and time-dependent effects

Twenty-four species exhibited supported overall or time-specific disturbance effects or notable changes in annual occupancy (including the 9 species highlighted above; Figs. 8, 9, 10). Black-backed Woodpecker, American Three-toed Woodpecker, Rock Wren, and Cassin’s Finch exhibited statistically supported shifts towards treated units in year 2 after prescribed fire (Fig. 8; see also similar but less supported pattern for Brewer’s Sparrow). Hermit Thrush shifted towards treated units in years 1 and 2, and then shifted back in year 3 following prescribed fire. Hammond’s Flycatcher, Chipping Sparrow, Ruby-crowned Kinglet, and Calliope Hummingbird exhibited lagged shifts towards untreated units in years 2 or 3 following prescribed fire. The 5 species with supported overall or time-dependent positive wildfire effects — Hairy Woodpecker, Olive-sided Flycatcher, House Wren, White-breasted Nuthatch, and Brewer’s Sparrow — all shifted towards higher severity burned points primarily in years 2–3 after wildfire (Fig. 9). Occupancy for the 3 species with overall negative wildfire effects (Cassin’s Vireo, Warbling Vireo, and Townsend’s Warbler; see Figs. 3, 5) shifted immediately towards lower severity burned points and remained there in all three years following wildfire (Fig. 9). Red-breasted Nuthatch, White-breasted Nuthatch, Hammond’s Flycatcher, Lazuli Bunting, and Pine Siskin initially shifted towards lower severity burned points in year 1 but then shifted back in years 2–3 after wildfire. Of particular note, White-breasted Nuthatch, Red-breasted Nuthatch, and Townsend’s Warbler shifted towards treated units after prescribed fire (Fig. 6). These prescribed fire effects translated into smaller changes in occupancy than did wildfire effects (compare Fig. 6 with Figs. 4, 5).
Dusky Flycatcher, Brewer’s Sparrow, MacGillivray’s Warbler, and Hermit Thrush also exhibited ephemeral shifts towards lower severity burned points in year 1, although these shifts received relatively weak statistical support.

Annual occupancy varied notably among pre-treatment (2002–2003), post-treatment (2004–2007), and post-wildfire years (2008–2010) for many species, providing further insight into disturbance effects (Fig. 10). In addition to shifting towards occupying high-severity burned points, Hairy Woodpecker and House Wren occupancy increased overall after wildfire. Mountain Bluebird, Dusky Flycatcher, Black-headed Grosbeak, Lazuli Bunting, MacGillivray’s Warbler, Cassin’s Finch, and Pine Siskin also exhibited notable albeit sometimes lagged increases in occupancy (Fig. 10) despite weakly supported or negative shifts in occupancy with respect to burn severity following disturbance (Figs. 8, 9). Hammond’s Flycatcher and Hermit Thrush exhibited declines in occupancy following disturbance (Fig. 10).

**Species richness**

We observed no definitive effects of prescribed fire or wildfire on species richness. Species richness increased overall by ~ 5 species following wildfire, but 95% BCIs for site-specific richness estimates overlapped considerably (Fig. 11 top row). Additionally, high-severity burned sites did not become any more or less species rich than lower severity sites following wildfire (i.e., a slight negative relationship with CBI was maintained; Fig. 11 top row). Overall species richness did not substantially change nor did the difference in species richness between treated and untreated units following prescribed fire (Fig. 11 bottom row).

**Discussion**

Our results suggest prescribed fire does not necessarily change the short-term ecological importance of wildfire to birds, even with limited wildfire burn severity. Prescribed fire limited subsequent wildfire burn severity within treated units (see also Pollet and Omi 2002; Prichard and Kennedy 2014; Fernandes 2015; Cary et al. 2017) but did not substantially modulate avian responses to wildfire. Species exhibiting the strongest shifts in occupancy towards or away from high-severity burned points after wildfire were similar in units initially treated versus not treated with prescribed fire.

Our study also highlights the limited ability of prescribed fire to emulate wildfire effects on birds. Wildfire substantially and strongly affected shifts in occupancy in relation to burn severity for multiple bird species. Differing metrics prevented us from quantitatively comparing wildfire with prescribed fire effects. Nevertheless, prescribed fire qualitatively affected a different and smaller set of bird species than did wildfire. Several species exhibited dramatic changes in overall occupancy following wildfire but not prescribed fire. Prescribed fire effects tended to be relatively time dependent and brief compared to wildfire effects. Finally, prescribed fire treatments would probably not extend across areas comparable to those burned by wildfire.

Wildfire affected communities primarily by altering the distribution of individual species rather than overall richness. Species richness increased somewhat after wildfire, but the magnitude of this change was small compared to variation among sites burned at similar severity, and wildfire did not substantially change which sites were most speciose. Rather, wildfire effects were primarily apparent for individual species modulated by life history (discussed further below), implying changes in species composition. Thus, the significance of wildfire to bird communities and avian diversity depends on the extent, distribution, timing, and severity of wildfire across landscapes (Kalies et al. 2010; Fontaine and Kennedy 2012; Latif et al. 2016b).

**Fire effects depend on species life history, population response, and resource dynamics**

Disturbance effects on species occupancy were generally consistent with species life histories and patterns reported in the literature (Smucker et al. 2005; Kotliar et al. 2007; Russell et al. 2009; Fontaine and Kennedy 2012; Latif et al. 2016b). Positive fire effects (i.e., overall or time-dependent shifts towards burned sites after wildfire or prescribed fire) on bark-drilling woodpeckers (Hairy, American Three-toed, and Black-backed Woodpecker) were congruent with their reliance on standing dead wood for nesting, and bark (e.g., Scolytidae) and wood-boring (Cerambycidae and Buprestidae) beetle larvae for food (Covert-Bratland et al. 2006; Kotliar et al. 2008). More pronounced prescribed fire compared to wildfire effects for disturbance specialist Black-backed and American Three-toed Woodpecker were unexpected, but sample sizes for these species were low and a lack of unburned sites (CBI = 0) may have limited power for estimating wildfire effects. Data describing specific life activities (e.g., nesting, foraging, and dispersal) may more effectively resolve relationships with wildfire for these species (Kotliar et al. 2008; Saab et al. 2009; Latif et al. 2013).

Additional species with positive fire effects included secondary cavity nesters (House Wren, White-breasted Nuthatch), aerial insectivores (Olive-sided Flycatcher), and species that nest or forage in the understory or on the ground (Brewer’s Sparrow, Rock Wren). These effects likely reflect increased nesting opportunities in snags generated by wildfire (secondary cavity nesters), increased foraging opportunities in canopy openings (aerial insectivores), and improved habitat quality with understory vegetation (understory species) (Kotliar et al. 2002; Saab et al. 2005; Smucker et al. 2005; Fontaine and Kennedy 2012; Latif et al. 2016b). Species exhibiting negative fire effects (shifts away from burned sites) included open-cup canopy nesters and bark- and foliage-gleaning insectivores (Mountain Chickadee, Cassin’s Vireo, Warbling Vireo, and Townsend’s Warbler), reflecting expected net losses in resources for these species (Kotliar et al. 2002; Saab et al. 2005; Smucker et al. 2005; Fontaine and Kennedy 2012; Latif et al. 2016b).

Time-dependent fire effects and overall changes in occupancy suggest some potential nuances in population responses or how fire affects resources. For bark-drilling woodpeckers, wood-boring beetle prey primarily colonize burned forests in year 2 following lags in tree mortality (Ray et al. 2019). Increased cavity availability for secondary cavity nesting species follows excavation by woodpeckers in initial years (Norris and Martin 2010). More generally, lagged positive effects may reflect greater recruitment of young in subsequent years produced initially by a relatively small number of burned-site colonists or residents. Conversely, site fidelity in the early postfire years may delay negative population responses for species reliant on green foliage for nesting and foraging.

Following wildfire in temperate forests, soil nutrient releases are typical and herbaceous vegetation regrowth begins within 1 year (Boemer 1982). Shrubs tend to dominate for the next 5–6 years (Schlesinger and Gill 1980; Boemer 1982), a process benefiting a variety of avian species (e.g., Hannon and Drapeau 2005;
Saab and Powell 2005; Fontaine and Kennedy 2012). In contrast, brief positive effects of prescribed fire suggest relatively short-lived resource pulses for affected species (e.g., Cassin's Finch and Hermit Thrush).

Resources may be greatest initially at lower severity burned sites after wildfire for species that nest or forage in the understory or on the ground. As time since fire progresses, however, resource availability may increase with increasing productivity at high-severity sites. Overall increases in occupancy after wildfire coupled with short-lived negative effects followed by positive effects in subsequent years suggested such resource dynamics for some species (e.g., Lazuli Bunting and Pine Siskin).

**Study limitations and future directions**

We did not include prescribed fire or wildfire effects on detectability, so fire effects on occupancy reported here do not explicitly control for potential heterogeneity in detectability. In preliminary analyses, we did not find statistical support for such effects (Q. Latif and V. Saab unpublished data), potentially reflecting a lack of such effects or limitations in statistical power. Given our sampling design, however, detectability could include information on movement between replicate surveys (Latif et al. 2016a) or heterogeneity in abundance (Royle and Nichols 2003), both of which are ecologically relevant. By ignoring fire effects on detectability, we forced any fire effects on movement or local abundance to be reflected in occupancy rather than detectability estimates (sensu Latif et al. 2018). Nevertheless, further study of fire or habitat effects on observer error could provide additional insight for interpreting our results.

Unlike others (Russell et al. 2009), we did not include persistence effects relating occupancy with the prior year's occupancy state (see also Russell et al. 2015). Having used BACI to control for potentially confounding spatial and temporal variation, we sought to avoid including further complexity and maximize information for estimating disturbance effects. Future study employing models representing occupancy dynamics (e.g., colonization, persistence, turnover) may yield additional insights into mechanisms underlying patterns observed here.

Our results are limited to fuels reduction treatments consisting exclusively of prescribed fire of primarily low severity in dry conifer forests characterized by mixed severity fire regimes. Birds have been shown to respond more strongly to prescribed fire treatments elsewhere likely due to greater treatment severity (e.g., Russell et al. 2009; Bagne and Purcell 2011). Greater treatment severity may elicit responses that more closely resemble responses to wildfire, but such treatments may also alter behavior and ecological significance of subsequent wildfire. Selective timber harvest may further alter direct and indirect treatment implications for birds by removing substantial standing woody biomass (Sallabanks et al. 2000; Perry and Thill 2013). Direct and indirect ecological implications of fuels treatments may further depend on historical fire regime (sensu Latif et al. 2016b). Studies examining interactions of fuels treatments, wildfire, and birds across a range of treatment severities, types, and sizes are needed to fully inform management of fire-adapted forests with objectives that include conservation of avian diversity.

**Management implications**

Our results suggest managers can use prescribed fire to limit burn severity of subsequent wildfire without completely compromising the value of wildfire to fire-associated species (e.g., woodpeckers, secondary cavity nesting species, aerial insectivores, and understory species) within burned areas. Given similar species relationships with burn severity in areas previously treated with prescribed fire (as observed here), we expect wildfire to generate some habitat for fire-associated species, albeit potentially less than in untreated areas in so far as burn severity is limited. In contrast, prescribed fire is unlikely to resemble wildfire in its ecological value for fire-associated species, limiting the value of prescribed fire as a surrogate for wildfire. Instead, the implications of prescribed fire for avian diversity in dry conifer forests may hinge more so on whether and how it shapes the spatial extent of subsequent wildfire. Where treated units are dwarfed by wildfire (e.g., Fig. 1), treatments may never be extensive enough to limit potential wildfire severity across entire landscapes. Nevertheless, prescribed fire treatments arranged strategically could break up landscapes and limit wildfire spread, particular in conjunction with other fuel treatments and fire control measures (Ackle et al. 2012; Hunter and Robles 2020). In multiple use forests, such fire management strategies would ideally allow sufficient wildfire to maintain biodiversity while limiting wildfire extent enough to meet other objectives (e.g., human safety, infrastructure protection).

**Conclusions**

We leveraged a rare opportunity to study serial effects of prescribed fire and subsequent wildfire on small landbirds. To the extent that wildfire affects resources for other species similarly, our study provides evidence that prescribed fire does not necessarily compromise the ecological value of subsequent wildfire for wildlife. Conversely, prescribed fire could help counteract effects of climate warming by limiting burn severity. To the extent that climate warming compromises the ecological value of wildfire by increasing its extent and severity (e.g., Jones et al. 2021), effective application of prescribed fire could help mitigate these impacts.

**Declarations**

**Ethics approval and consent to participate**

Not applicable

**Consent for publication**

Not applicable

**Data availability**

The datasets used and/or analyzed here are available from the corresponding author on reasonable request.
Competing interests

The authors declare that they have no competing interests

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Author contributions

VAS designed the study and obtained funding. VAS and JGD organized and oversaw data collection. QSL and VAS developed the analysis approach. QSL implemented the analysis and drafted the manuscript. VAS and JGD contributed editorial input during manuscript preparation.

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Figure 1

Study area and units where forest bird data were collected in the Payette National Forest (Idaho, U.S.A.) in relation to prescribed fire treatments and subsequent wildfire. The fire perimeter is for the 2007 East Zone Complex wildfire.
Figure 2

Box plots depicting distributions of composite burn index (CBI) values for survey points treated by prescribed fire and burned by wildfire. Wildfire CBIs are shown separately for units previously treated versus not treated with prescribed fire. Boxes delineate the 25th, 50th, and 75th percentiles, whiskers denote distance to observations furthest from the nearest quartiles (i.e., 25th and 75th percentiles) that are also within 1.5 × the inter-quartile range from the nearest quartile, and dots are observations further than 1.5 × the inter-quartile range from the nearest quartile.
Posterior median estimates (dots) with 90% credible intervals (error bars) for wildfire and prescribed fire effects on avian species occupancy at the Payette National Forest (Idaho, U.S.A.). Full species names are provided in Table 2. Error bars are color coded based on statistical support (credible intervals excluding zero) and direction (orange = positive; blue = negative). Positive versus negative values indicate occupancy shifts towards versus away from (respectively) treated (prescribed fire treatment) or high-severity burned (wildfire CBI) sites. $\beta_{\text{wildfire CBI}}$ represents $\beta_{\text{perWFxCBIWF}}$ in Equation 4. $\beta_{\text{prescribed fire treatment}}$ represents $\beta_{\text{perPFxTrtWF}}$ in Equation 3.
Figure 4

Predicted occupancy probabilities along a wildfire burn severity gradient (composite burn index; CBI) for species exhibiting statistically supported positive wildfire effects, i.e., where occupancy shifted towards high-severity burned sites after wildfire. Occupancy relationships are depicted before (gray) and after (black) wildfire. Species are House Wren (HOWR), Hairy Woodpecker (HAWO), Olive-sided Flycatcher (OSFL), and Brewer's Sparrow (BRSP).
Figure 5

Predicted occupancy probabilities along a wildfire burn severity gradient (composite burn index; CBI) for species exhibiting statistically supported negative wildfire effects, i.e., where occupancy shifted towards low-severity or unburned sites after wildfire. Occupancy relationships are depicted before (gray) and after (black) wildfire. Species are Cassin’s Vireo (CAVI), Townsend’s Warbler (TOWA), and Warbling Vireo (WAVI).

Figure 6

Predicted occupancy probabilities for treatment versus control units before (gray) and after (black) prescribed fire for species exhibiting statistically supported prescribed fire effects, i.e., where occupancy shifts towards or away from treated sites. Species depicted are American Three-toed Woodpecker (ATTW) and Rock Wren (ROWR).
Figure 7

Posterior median estimates (dots) with 90% credible intervals (error bars) for wildfire effects on avian species occupancy at the Payette National Forest (Idaho, U.S.A.). Positive versus negative values indicate occupancy shifts towards versus away from (respectively) high-severity burned sites after wildfire in units previously treated with prescribed fire. Full species names are provided in Table 2. Error bars are color coded based on statistical support (credible intervals excluding zero) and direction (orange = positive; blue = negative). \( \beta \) Wildfire CBI (untreated) represents \( \beta_{\text{PerWF} \times \text{CBIWF},j} \) and \( \beta \) Wildfire CBI (treated) represents \( \beta_{\text{PerWF} \times \text{CBIWF},j} + \beta_{\text{PerWF} \times \text{TrtPF} \times \text{CBIWF},j} \) in Equation 5.
Figure 8

Posterior median estimates (dots) with 90% credible intervals (error bars) for time-dependent prescribed fire effects on avian species occupancy at the Payette National Forest (Idaho, U.S.A.). Estimates are for all 24 species with statistically supported time-dependent disturbance effects or notable changes in annual occupancy overall (see full species names in Table 2). Error bars are color coded based on statistical support (credible intervals excluding zero) and direction (orange = positive; blue = negative). Positive versus negative values indicate occupancy shifts towards versus away from (respectively) treated sites after treatment. βPrescribed fire treatment represents \( \beta_{\text{TrtPF}_i(t_{PF}=1,2,3)} - \beta_{\text{TrtPF}_i(t_{PF}=0)} \) in Equation 6, where \( t_{PF} = 1, 2, \) and 3 correspond with year 1 (Y1), year 2 (Y2), and year 3 (Y3), respectively, after prescribed fire.
Figure 9

Posterior median estimates (dots) with 90% credible intervals (error bars) for time-dependent wildfire effects on avian species occupancy at the Payette National Forest (Idaho, U.S.A.). Estimates are for all 24 species with statistically supported time-dependent disturbance effects or notable changes in annual occupancy overall (see full species names in Table 2). Error bars are color coded based on statistical support (credible intervals excluding zero) and direction (orange = positive; blue = negative). Positive versus negative values indicate occupancy shifts towards versus away from (respectively) high-severity burned sites after wildfire. $\beta_{\text{Wildfire CBI}}$ represents $\beta_{\text{CBIWF}_i(t_{\text{WF}} \in \{1,2,3\})}$ - $\beta_{\text{CBIWF}_i(t_{\text{WF}}=0)}$ in Equation 6, where $t_{\text{WF}}$ = 1, 2, and 3 correspond with 2008 (Y1), 2009 (Y2), and 2010 (Y3).
Figure 10

Posterior median estimates (dots) with 90% credible intervals (error bars) for annual occupancy of surveyed sites for bird species at the Payette National Forest (Idaho, U.S.A.). Estimates are for all 24 species with statistically supported time-dependent disturbance effects or notable changes in annual occupancy overall (see full species names in Table 2). The vertical dashed lines demark when prescribed fire treatments were applied (one unit in 2004; 2 units in 2006), and the vertical solid line demarks when wildfire occurred.
Figure 11

Species richness posterior estimates (median with 90% BCIs) for point × year survey occasions plotted against wildfire burn severity (CBI; top panels) and within prescribed fire treatment versus control units (bottom panels) before (left) versus after (right) disturbance. Best-fit lines show mean species richness trends for posterior median estimates.

Supplementary Files

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