Before-after-control-impact study reveals muted avian response to low-severity wildfire

by
Lucy Heflin

A THESIS

submitted to

Oregon State University

Honors College

in partial fulfillment of
the requirements for the
degree of

Honors Baccalaureate of Science in Natural Resources
(Honors Scholar)

Presented October 23, 2025
Commencement June 2026

AN ABSTRACT OF THE THESIS OF

Lucy Heflin for the degree of Honors Baccalaureate of Science in Natural Resources presented on October 23, 2025. Title: Before-after-control-impact study reveals muted avian response to low-severity wildfire.
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Abstract approved:____

Matthew Betts

Wildfire is increasing in frequency and severity across the western United States, highlighting a need to understand how biological communities are responding to this disturbance. Bird communities are useful indicators of ecological change as they are relatively easy to survey and are a well-studied taxonomic group in terms of behavior and life history. I utilized a before-after-control-impact study design to test three hypotheses regarding bird response to wildfire in the mixed-conifer forests of the H.J. Andrews Experimental Forest, Blue River, Oregon following the Holiday Farm Fire of 2020. The *Vegetation Selection Hypothesis* states that after fire, birds will move to forests that most closely resemble their preferred seral stage; the *Site Fidelity Hypothesis* states that after fire, birds will return to where they occurred the previous year; and the *Concussion Effect Hypothesis* states that the landscape effect of fire will lead to birds moving into nearby, unburned habitat. Point count data collected two years prior to the fire and three years after were used to model changes in total abundance for six focal bird species. I did not detect a strong effect of fire on most of the species in my study. However, the Western Flycatcher (*Empidonax difficilis*) did have abundance changes that showed moderate support for the *Concussion Effect Hypothesis* Due to the small spatial replication in this study, I recommend further research into bird responses to wildfire using a larger dataset that includes varying fire severity.

Keywords: wildfire, birds, abundance, site fidelity, vegetation selection, dispersal

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Honors Baccalaureate of Science in Natural Resources project of Lucy Heflin presented on October 23, 2025.

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I understand that my project will become part of the permanent collection of Oregon State University, Honors College. My signature below authorizes release of my project to any reader upon request.

Lucy Heflin, Author

1. INTRODUCTION

The western United States is experiencing an increase in the frequency and severity of wildfires due at least partly to anthropogenic climate change (McKenzie et al. 2004; Westerling et al. 2006). Increased temperatures in the spring and summer, lower winter precipitation, and earlier spring snowmelt are creating longer summer seasons with drier vegetation; this not only extends the fire season but is also driving larger and longer burning fires (McKenzie et al. 2004; Westerling et al. 2006). As wildfire disturbance events increase in frequency, it becomes important to understand their impacts on forest landscapes. In Oregon, 20 wildfires burned nearly simultaneously across the state in the Labor Day fires of 2020, highlighting the impacts of extreme meteorological conditions on fire activity (Abatzoglou et al. 2021). Strong easterly winds, coupled with antecedent hot and dry weather in July and August led to extreme fuel dryness during fire events in early September. This resulted in a fire complex that burned 393,315 hectares of forest (Abatzoglou et al. 2021). Overall, the 2020 fire season resulted in more than 11% of the Cascade Mountain ecoregion in Oregon being burned (Abatzoglou et al. 2021).

With such a large amount of forest burned, ecologists have an interest in understanding species' ability to adapt to changing landscapes, as well as how biological communities reassemble following fire. Birds have been used as indicators of ecological change for several reasons – they are easy to identify, inexpensive to survey, and there is substantial available information on species and their basic behaviors and life history (Niemi and McDonald 2004). Birds are also the focus of many long-term monitoring programs, so there are more data to help distinguish between abundance changes due to a disturbance versus baseline population dynamics (Canterbury et al. 2001). Previous research has shown that changes in bird communities can coincide with changes in forest structure due to disturbance (O'Connell et al. 2000; Canterbury et al. 2001). Investigating bird communities' response to wildfire in their breeding areas may give more insight into how landscape changes are impacting forest communities.

Forest structure refers to the distribution of both live and dead vegetation in a forest ecosystem and commonly shifts following wildfires, although the degree of change depends on the severity of the fire (Spies 1998). Low-severity fire typically burns on the surface, producing mortality and altering structure within the understory but has very minimal effects on the overstory (Perry et al. 2011). After a high-severity, stand-replacing wildfire, forests in the Pacific Northwest (PNW) are often colonized first by grasses and herbs, then eventually dominated by woody shrubs and hardwood. These forests also tend to have a large number of snags, and dead and downed wood (Swanson et al. 2014). Recently burned forests are classified as "early seral" and provide important habitat for certain species groups, including birds (Betts et al. 2010; Swanson et al. 2014). This is specifically true of high-severity burned forests, as low-severity burned forests often don't experience enough change to the vertical forest structure and overall composition to alter the successional stage (Perry et al. 2011). Specifically, the broadleaf trees and shrubs characteristic of this forest type support a higher abundance of moths and butterflies, a common food source among birds (Hammond and Miller 1998), and thus are known to be associated with high abundances of common PNW bird species (Betts et al. 2010). Coniferous Douglas-fir (*Pseudotsuga menziesii*) forests take three to four decades post-fire to fully establish themselves and begin to close canopy; early seral forest could persist for multiple decades and provide important, structurally complex habitat for birds that are associated with this forest type (Franklin et al. 2002; Betts et al. 2010; Donato et al. 2011; Tepley et al. 2014). Ecologists refer to the vegetative species that make up forest ecosystems as "forest composition", which tends to shift following wildfire. This can bring about new habitat and food sources for forest-associated species, leading to species-specific responses to wildfire.

Among birds, there are several mechanisms that could drive individual species responses to wildfire. First, migrating songbirds are known to develop attachment to specific breeding sites and return year after year (Greenwood and Harvey 1982). This process, termed "site fidelity", is hypothesized to have evolved due to the benefits associated with habitat familiarity (Greenwood and Harvey 1982). When birds are more familiar with the location of food sources within their habitat, they can collect resources more efficiently (Greenwood and Harvey 1982). Familiarity with their habitat also provides an advantage in aggressive encounters with other birds (Greenwood and Harvey 1982). Finally, because nesting resources and predation risk tend to be temporally autocorrelated, birds are likely to return to locations where they have been successful at breeding in previous years (Danchin et al. 2004). After a wildfire disturbance, site fidelity could contribute to temporal lags in abundance changes of a species.

Second, in landscapes where habitat has been lost due to a disturbance, birds may move to nearby habitat that has not experienced this disturbance, a phenomenon termed the "concussion effect" (Hagan et al. 1996; Betts et al. 2006). Disturbance and subsequent habitat fragmentation can be caused by natural and human-driven processes including wildfire, although the latter has not been quantitatively demonstrated. Movement from lost habitat is likely to occur on a relatively fine spatial scale due to benefits of local habitat knowledge and dispersal limitations (Betts et al. 2006). The displacement from lost habitat may cause increased bird density and potentially crowding in areas close to recently disturbed habitat (Hagan et al. 1996).

In this study, I investigated how birds respond to the disturbance of wildfire. I first hypothesized that birds would settle where they have their preferred habitat (*Vegetation Selection Hypothesis*). If this hypothesis is true, I would expect to see birds that have an early seral habitat association increase in abundance immediately post-fire, while birds that have mid- and late-seral habitat associations would be expected to decline in abundance immediately post-fire. Alternatively, I hypothesized that birds would return to where they occurred the year before the fire (*Site Fidelity Hypothesis*). If this hypothesis is true, I would predict to see no change in the abundance of bird species in the year immediately following fire. My third hypothesis is that fire has a landscape-scale impact on unburned areas that were adjacent to the burn, and that movement will occur – possibly driven by the *Vegetation Selection Hypothesis* – into nearby, unburned areas in years following the immediate post-fire years, boosting density in these locations (*Concussion Effect Hypothesis*). If the concussion effect hypothesis is true, I would expect to see a decrease in post-fire abundance in the burned areas and an increase in abundance in the adjacent, unburned areas. Unburned areas that are not adjacent to the fire should not experience a change in abundance.

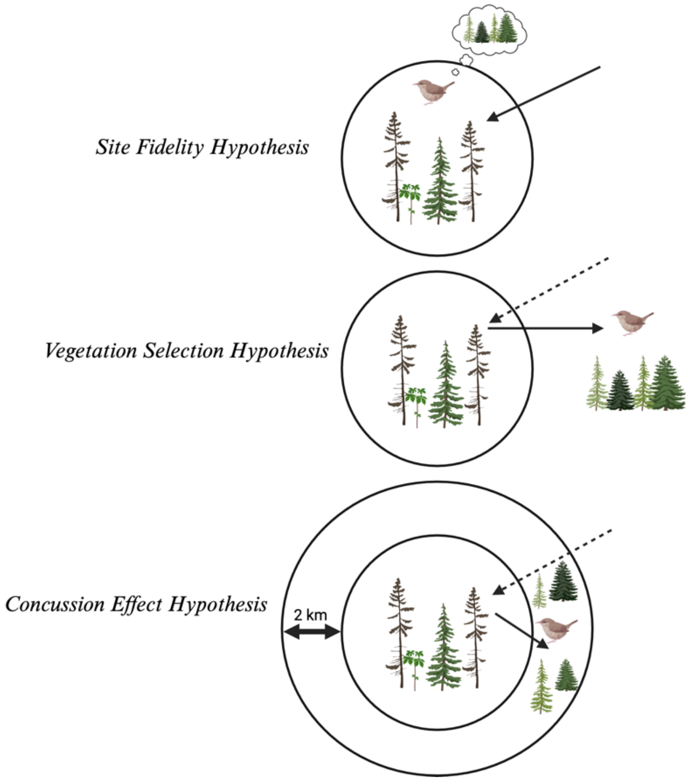

Figure 1. Conceptual diagram of bird movement under the *Site Fidelity Hypothesis*, *Vegetation Selection Hypothesis*, and *Concussion Effect Hypothesis*

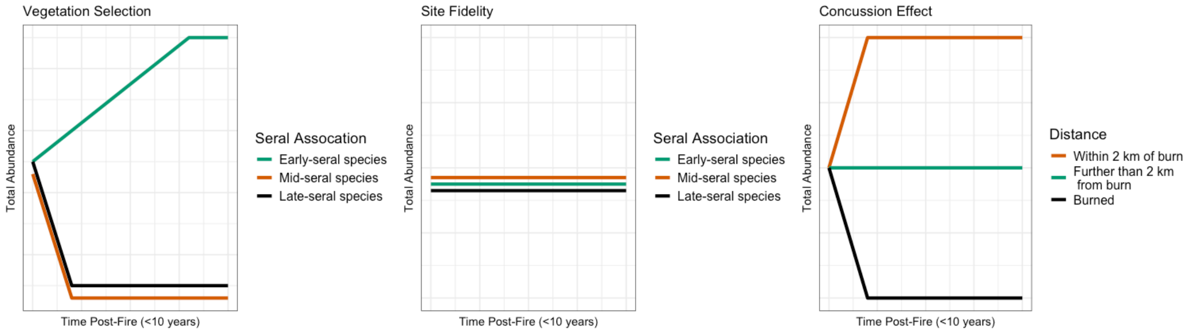

Figure 2. Predictions about the bird abundance over time since high-severity fire. The *Vegetation Selection Hypothesis* suggests birds will settle where they have their preferred habitat, the *Site Fidelity Hypothesis* suggests birds will return to where they occurred the year previous to the fire, and the *Concussion Effect Hypothesis* suggests birds will move into nearby, unburned areas in the years immediately post-fire.

2. METHODS

2.1 Study Site

This study used avian point count data collected at the H.J. Andrews Experimental Forest (hereafter ‘H.J.A.’; HJA 2020), a 6,400 hectare forest in the western Cascade mountains in Oregon. The H.J.A. encompasses a topographically diverse landscape, with elevation ranging from 410 to 1,630 meters. The forest is a mixture of old-growth and second growth stands, composed mainly of Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*) and western red cedar (*Thuja plicata*). Historically there have been a variety of disturbances, including wind, fire, ice and snow storms, landslides, and logging, which have produced a diversity of stand ages ranging from 30-700 (HJA 2020).

2.2 Point Count Data

Following the protocol outlined in Kim et al. (2022), we collected avian point count data at 46 sites across the H.J.A. We surveyed each point between three and six times during the breeding season of songbirds (May 1 – July 15) each year, ensuring at least five to seven days between each replication (Kim et al. 2022). We only conducted surveys between 5:15 AM and 10:30 AM under conditions of good visibility, little or no precipitation and little or no wind (Kim et al. 2022). We recorded all bird species heard or seen during this time (Kim et al. 2022).

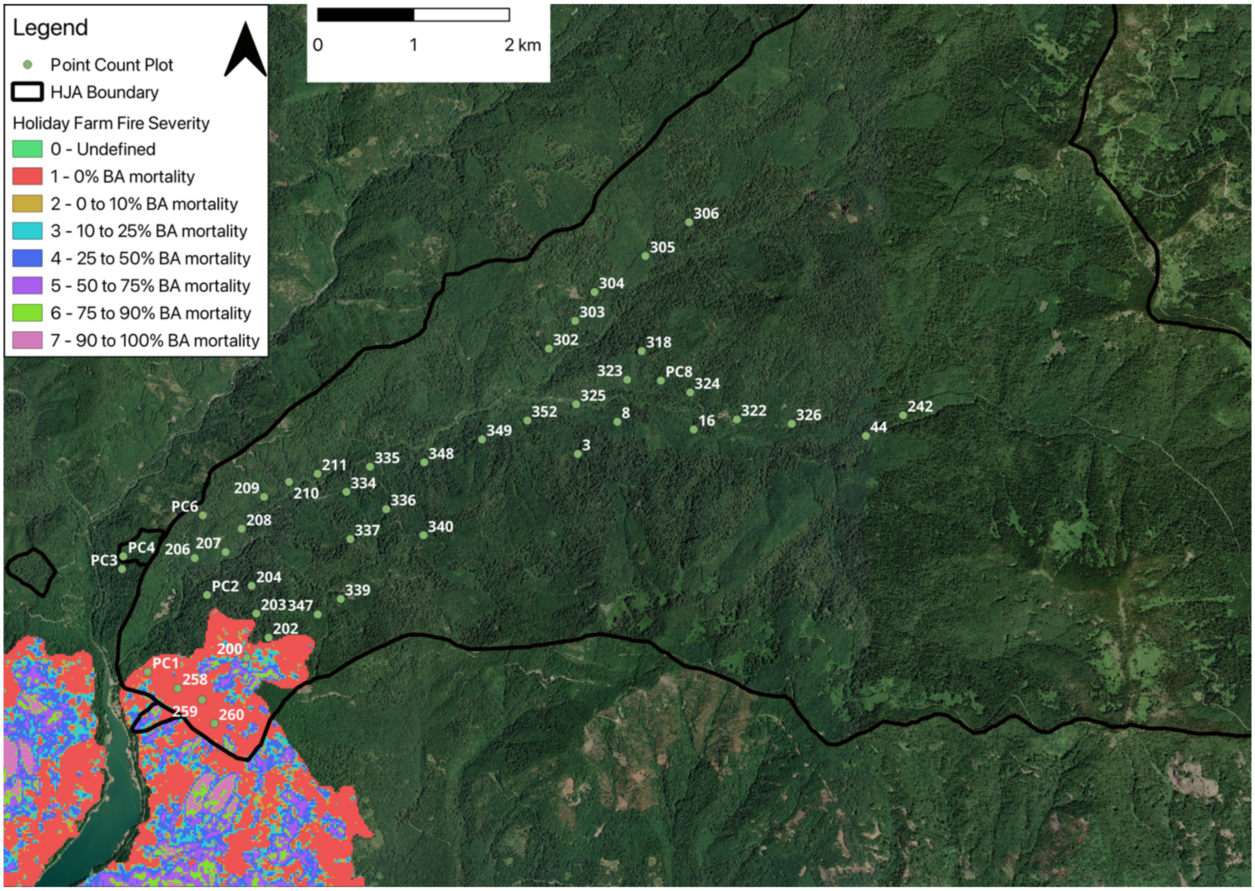

Figure 3. Location of study plots within the H.J.A. overlayed on the Holiday Farm Fire perimeter and severity map. Basal area (BA) is a measure of tree density and BA mortality is refers to the percentage of basal area killed by wildfire.

I used a before-after-control-impact study design to test the effect of wildfire on bird species total abundances. The "impact" in this study was the Holiday Farm Fire, which ignited on September 7, 2020 in the western Cascade mountains near the H.J.A. (USDA 2020). This fire burned over 173,000 acres, including about 400 acres of forest within the H.J.A. boundary (USDA 2020; HJA 2020). I used data from point counts conducted two years before the fire ("pre-fire") and three years after the fire ("post-fire"). Point count data from these years consisted of six points surveyed each year within the burned area and 39 points surveyed each year outside of the burned area, with 18 of those unburned points located more than two kilometers from the burn. I used the distance of two kilometers from the burn because it is the distance within which adjacent habitat may be experiencing effects of a disturbance, as well as the likely breeding dispersal distance of birds (Laurance 2000; Betts et al. 2006; Ewers et al. 2007). I calculated total abundance for each point by adding up the number of detections within 50 meters of the point for each focal species across all three visitations in a year. Then, I added up yearly totals to obtain a pre-fire and post-fire total abundance for each point for a given species. Lastly, I replicated the process shown below for each focal species. The values calculated for total abundance are likely overestimating the actual abundance of birds at a given point. It is possible that the same individual could be present during each visitation to a point in a given year; however, by adding up total abundance, we are making an assumption that this never happens which is unlikely. The value for total abundance should be treated as an index representing the abundance of a species, and not necessarily an accurate value of actual abundance. Total abundance was necessary to use in this study as opposed to mean abundance because the data was modeled using a negative binomial or Poisson distribution, both of which require count data. In these equations v represents abundance totals at a given visitation to a point, $t_{y,i}$ represents abundance total at a given point for a given year, and $T_{pre/post,i}$ represents pre and post fire abundance totals for a given point for a certain species.

$$t_{y,i} = v_1 + v_2 + v_3$$

$$T_{pre,i} = t_{2018,i} + t_{2019,i}$$

$$T_{post,i} = t_{2021,i} + t_{2022,i} + t_{2023,i}$$

Pre-fire data comprised 2018 and 2019 point count data, while point count data from 2021-2023 comprised the post-fire data. The fire burned five out of the six points in the burned area at very low severity (0% basal area mortality), meaning there was no change in live basal area after the fire. One point was burned at low/moderate severity (10-25% basal area mortality). I identified the burned points to be between 450 meters and 750 meters in elevation and selected this study's 39 unburned point count points to also be between 450 meters and 750 meters in elevation.

I used raw abundance data in this study, as correcting for potential detection bias by doing occupancy modeling was beyond the scope of this study (MacKenzie et al. 2017) and likely challenging due to low sample size. Although detection bias is present in nearly all avian point count surveys, I attempted to minimize bias without doing a statistical correction for it (Simons 2009). Point count surveys were deliberately designed to decrease observer effects (each of three visitations in a year were conducted by a different surveyor) and effects from time of day (points were surveyed in reverse order with each visitation to ensure they were surveyed at different times of the day). I filtered the data to only include birds recorded within 50 meters of the observer in the analysis; beyond this distance, detectability substantially declines (Shieck 1997). The lack of a large, leafy understory at the burned points may also decrease detection bias in the burned area; Marten and Marler (1977) found that for most frequencies, sound travels further in open habitat as opposed to leafed out forests. This bias toward observing increased abundances in burned versus unburned sites constitutes a conservative bias with respect to detecting change in late-seral bird species, but a potentially liberal bias for early seral species. By minimizing detection bias to the extent possible, I believe results will still be robust enough to detect abundance changes if they should happen.

I examined six bird species in this study, which I chose based on predicted habitat associations and their sufficiently high prevalence of the points used in this study (higher prevelance meant larger sample sizes). Black-throated Gray Warblers (*Setophaga nigrescens*) were used as the early seral associated species for this study (Phalan et. al 2019). Chestnut-backed Chickadees (*Poecile rufescens*), and Swainson's Thrushes (*Catharus ustulatus*) were used as the mid-seral associated species in this study (Harris and Betts, 2021). Hermit Warblers (*Setophaga occidentalis*), Pacific Wrens (*Troglodytes pacificus*) and Western Flycatchers (*Empidonax difficilis*) were used as the late-seral associated species (Harris and Betts 2021).

2.3 Statistical Analysis

To model total bird abundance under the *Vegetation Selection Hypothesis* and *Site Fidelity Hypothesis*, I used a generalized linear mixed model with a negative binomial distribution. This allowed me to account for both fixed and random effects, as well as handle the data's overdispersion. Point count plot was treated as a random factor in this model because surveys conducted at the same point across different years should be more alike than surveys conducted at different points and thus are not likely to be independent. Burn status (burned or unburned) and time (pre-fire or post-fire) were treated as fixed factors.

$$Y_{ij} = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_1 x_2 + \log(T) + b_i + \varepsilon_{ij}$$

In this model, the continuous variable Y_{ij} is the total abundance of a given bird species at point i in time period j. The variable x_1 identifies the time period (pre-fire or post-fire) and the variable x_2 identifies the treatment of point i (burned or unburned). The model intercept is shown with β_0, and can be interpreted as the estimated total bird abundance pre-fire at the unburned points. The coefficient β_1 represents the estimated change in total abundance at all points in the post-fire time period as compared to the pre-fire time period. The coefficient β_2 represents the estimated difference in total abundance at burned points compared to unburned points across all time periods. The coefficient β_3 represents the interaction between time period (x_1) and treatment (x_2) in predicting total abundance and can be interpreted as the estimated change in total abundance at the burned points post-fire. The log-transformed term T adjusts the expected total abundance based on the number of surveys done in the time period (either 2 years or 3 years' worth of surveys). This attempts to ensure that survey effort is proportional to expected total abundance. The variation among points is represented by the random effects term b_i and the residual variation is represented with the error term ε_{ij}.

To model data for the *Concussion Effect Hypothesis,* I again used a generalized linear mixed model that followed a negative binomial distribution (except in two instances as indicated with an asterisk in Table 2 where a Poisson distribution better fit the data). Point count point was still treated as a random effect. Point count points within the fire perimeter were not included in this dataset, and distance from burn (within 2 km of burn or further than 2 km from the burn) and time (pre-fire or post-fire) were treated as fixed factors.

$$Y_{ij} = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_1 x_2 + \log(T) + b_i + \varepsilon_{ij}$$

In this model Y_{ij} is the total abundance of a given bird species at point i in time period j. The variable x_1 identifies the time period (pre-fire or post-fire) and the variable x_2 identifies the distance from burn (further than 2 km from the burn or within 2 km of the burn). The coefficient β_3 represents the interaction between time period (x_1) and distance from burn (x_2) in predicting total bird abundance. The log-transformed term T adjusts the expected total abundance based on the number of surveys done in the time period (either 2 years or 3 years' worth of surveys) in an attempt to ensure that survey effort is proportional to expected total abundance. The variation among points is represented by the random effects term b_i and the residual variation is represented with the error term ε_{ij}.

3. RESULTS

3.1 Vegetation Selection/Site Fidelity Model

Overall, I found that none of the species I analyzed appeared to be affected by low-severity fire. I did not detect a statistically significant relationship to wildfire for any of the six species; effect sizes were small, confidence intervals broad, and p-values high.

Table 1. Results from the vegetation selection model measuring total bird abundance. Interaction term (β_3) indicates change in total bird abundance post fire within the burned landscape

Species	n	d.f.	$\hat{\beta}_3$	standard error	z-value	95% C.I.	p value
BTYW	46	40	-0.7596	0.6290	-1.2080	(-1.9925, 0.4733)	0.2270
CBCH	81	75	-0.3570	0.3882	-0.9200	(-1.1179, 0.4039)	0.3578
SWTH	85	79	0.4797	0.4281	1.1210	(-0.3594, 1.3188)	0.2625
PAWR	84	78	-0.4543	0.2703	-1.6810	(-0.9839, 0.0754)	0.0928
WEFL	80	74	-0.4499	0.3227	-1.3940	(-1.0823, 0.1826)	0.1633

HEWA	61	55	-1.0029	0.7183	-1.3960	(-2.4107, 0.0405)	0.1626

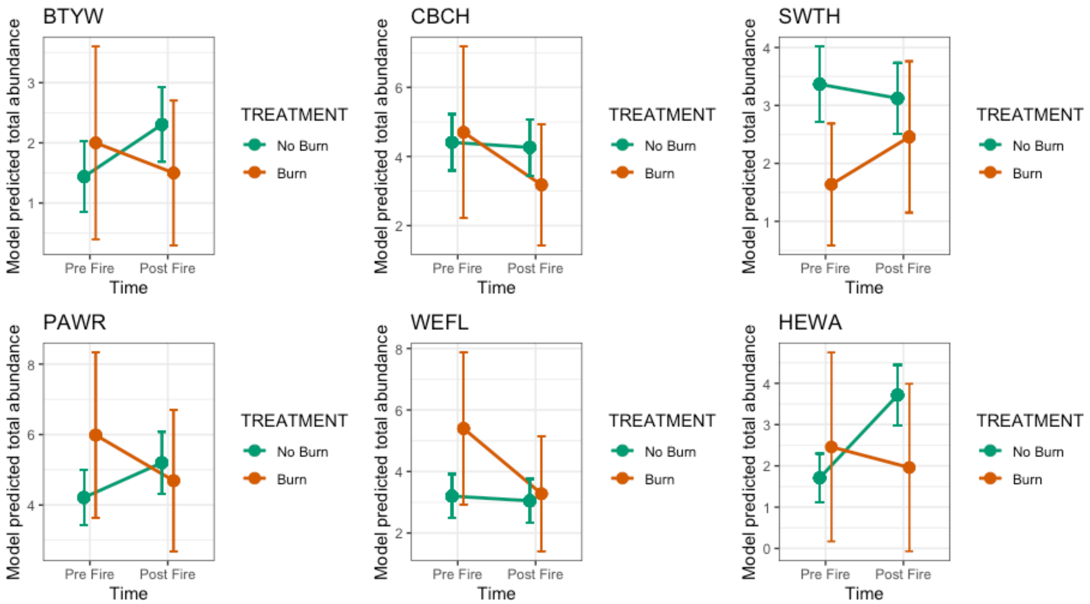

Figure 4. Model predicted total abundance for six bird species where pre-fire includes 2018-2019 data and post-fire includes 2021-2023 data. The “No Burn” treatment group included data points from varying distances within 2 kilometers and greater than 2 kilometers away from the burn.

3.2 Concussion Effect Model

Western Flycatchers tended to increase in abundance in areas surrounding the burn, which could be evidence for the concussion effect hypothesis. Abundance for this species increased by 0.79 (95% CI = 0.26 – 1.33 , p = 0.004) in the pre- to post-fire period as compared to the more distant unburned control sites. I did not detect a statistically significant relationship with distance from wildfire for any other species. All other species show varying relationships in total abundance change between pre- and post-fire periods depending on distance from the fire (Figure 5). None of these relationships appear to be strong; there are large confidence intervals and small effect sizes for nearly every species (Figure 5).

Table 2. Results from the concussion effect model which predicted abundance as a function of distance from the burn footprint; interaction term (β3) indicates change in total bird abundance post fire on the landscape within 2 kilometers of the burn. Asterisks (*) indicates species whose distribution was best fit with a Poisson distribution instead of a negative binomial.

species	n	d.f.	$\hat{\beta}_3$	standard error	z-value	95% C.I.	p
BTYW	39	33	-0.6221	0.5361	-1.1600	(-1.6728, 0.4287)	0.2460
CBCH	72	66	-0.0159	0.2550	-0.0620	(-0.5156, 0.4838)	0.9503
SWTH*	73	68	0.2471	0.2632	0.9390	(-0.2688, 0.7630)	0.3478
PAWR	72	66	0.0636	0.2151	0.2960	(-0.3579, 0.4851)	0.7675
WEFL*	**69**	**64**	**0.7945**	**0.2727**	**2.9130**	**(0.2600, 1.3290)**	**0.0036**
HEWA	57	51	-0.1154	0.3841	-0.3010	(-0.8682 0.6374)	0.7640

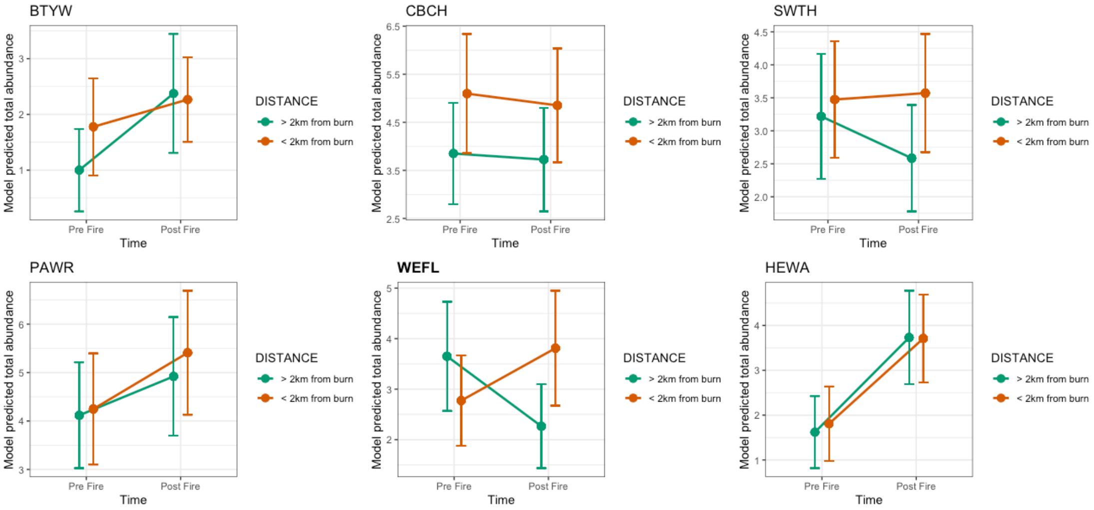

Figure 5. Model predicted total abundance for six bird species where pre-fire includes 2018-2019 data and post-fire includes 2021-2023 data. The "> 2 km from burn" group includes data from point counts conducted greater than two kilometers from the burn perimeter. The "< 2 km from burn" group includes data from point counts conducted within two kilometers of the burn perimeter but does not include any locations within the burn perimeter.

4. DISCUSSION

Overall, I found few statistically significant changes to total bird abundance post low-severity wildfire in a mixed-conifer forest. The small sample size of point counts within the burn perimeter, coupled with the relatively muted response by most species to low burn severity likely reduced the statistical power of our models. With such low statistical power, it is unlikely that these models could detect ecologically significant changes in abundance. This makes it unclear whether the lack of strong results is due truly to the lack bird dispersal post-fire or is rather due to the insufficient power of the models.

The *Vegetation Selection Hypothesis* states that birds should track their preferred vegetation structure and composition; thus, later-seral species should show declines following fire's disruption of habitat characteristics. The late-seral species analyzed in this study (Pacific Wren, Western Flycatcher, and Hermit Warbler) all did show slight declines in abundance following the low-severity wildfire (see Figure 4). However, with such a small sample size and large confidence, these decreases were not statistically significant. A low-severity fire has small, short-term effects on the ecosystem that are mainly concentrated in the understory; the overstory is often not affected (Agee 2007). This could explain why canopy foraging species, such as the Hermit Warbler, did not disperse as predicted in the *Vegetation Selection Hypothesis*. The short-term ecosystem effects from low-severity fire are not often strong enough to change the successional stage of the forest (Agee 2007), which could explain why there were no significant changes in abundance between pre-fire and post-fire for any species.

The *Site Fidelity Hypothesis* states that birds develop an attachment to the site they bred the previous year and will return there the next year regardless of disturbance. Under this hypothesis, there should be a time lag in bird abundance declines following fire. Large confidence intervals around estimates of bird abundance at the burned sites due to the small sample size made it difficult to detect whether a site fidelity effect occurred. It is possible that the small ecological effect of low-severity wildfire (Agee 2007) did not have a strong impact on bird dispersal, and site fidelity did continue to drive yearly dispersal. The slight abundance changes seen in my models could be reflective of natural fluctuations in yearly bird populations.

Under the *Concussion Effect Hypothesis*, abundance is expected to increase post-fire within two kilometers of the burn but stay relatively constant at distances greater than two kilometers from the burn. Western Flycatcher abundance tended to increase significantly in areas proximal to the burn, which is evidence for the *Concussion Effect Hypothesis* (see Figure 5). It is possible Western Flycatcher populations are decreasing across the landscape, as seen with the decrease in abundance beyond two kilometers from burn but increasing within the two-kilometer buffer zone due to the concussion effect. However, the decrease in abundance of Western Flycatchers far from the burn weakens the argument for the concussion effect because it suggests birds were also moving out the area or suffering mortality far from the burn. Additionally, the lack of substantial vegetation change following a low-severity fire could explain why there does not seem to be a stronger two kilometer concussion effect for more species. Birds may not make the effort to disperse to nearby refugia if their current habitat has not been greatly altered. It is important to note that the Holiday Farm fire was very large (>100,000 acres) with much of its footprint burned at medium (25%-75% basal area mortality) and high severity (>75% basal area mortality), although none of those areas were within the H.J.A. (USDA 2020). It is possible that the 400 acres of patchy, low-severity burn within the H.J.A. provided refugia for birds who normally utilized the now-burned habitat outside of the H.J.A.

5. CONCLUSION

My study aimed to provide a greater understanding of the ecological forces driving bird abundance after wildfire. Understanding the landscape-scale effect of fire on bird abundance could have implications for post-fire management and habitat protection. It could also help to inform silvicultural practices that better mimic natural disturbances, improving upon ecological approaches to forestry (Franklin et al. 2007). Given relatively low statistical power due mostly to low sample sizes, further research should be done on how bird abundance changes after wildfire, including how fire severity impacts these changes. There is opportunity for replication on a broader scale at the H.J.A. using point count data following the 2023 Lookout Fire, which was much larger and of mixed severity. Improvements to this study design would be most substantial by increasing sample sizes to enhance statistical power. This could be done by studying a larger burn that included more area within the long-term dataset. Additionally, a fire that burned with mixed severity across sample points would provide the opportunity to analyze bird responses across a severity gradient. I predict that species responses to fire should be much more pronounced with dramatic changes to vegetation structure and composition. However, it is possible that these effects may not manifest until two years after the fire, due to site fidelity (Greenwood and Harvey 1982). The Lookout Fire provides the opportunity to both increase sample sizes and to examine a wider range of fire severity. A third and perhaps more costly improvement to be made is the color banding of birds both within the burn and outside of the burn. This would confirm an individual bird's movement, something a point count is unable to accurately do. Wildfires burning at a greater frequency across the western U.S. both increase the need, and provide ample opportunity, for similar but potentially more significant studies around bird behavior after wildfire.

References

Abatzoglou, J. T., D. E. Rupp, L. W. O'Neill, and M. Sadegh. 2021. Compound Extremes Drive the Western Oregon Wildfires of September 2020. *Geophysical Research Letters* 48(8):e2021GL092520. https://doi.org/10.1029/2021GL092520.

Agee J. 2007. Low-severity fire. *In*: FireWords: Fire Science Glossary [electronic]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). http://www.fs.fed.us/fmi.

Balch, J. K., B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, and A. L Mahood. 2017. Human-started wildfires expand the fire niche across the United States. *PNAS* 114(11):2946-2951. https://doi.org/10.1073/pnas.1617394114.

Betts, M. G., B. P. Zitske, A. S. Hadley, and A. W. Diamond. 2006. Migrant Forest Songbirds Undertake Breeding Dispersal Following Timber Harvest. *Northeastern Naturalist* 13(4):531-536. https://www.jstor.org/stable/4130987.

Betts, M., J. C. Hagar, J. W. Rivers, J. D. Alexander, K. McGarigal, and B. C. McComb. 2010. Thresholds in forest bird occurrence as a function of the amount of early-seral broadleaf forest at landscape scale. *Ecological Applications* 20(8):2116-2130. https://doi.org/10.1890/09-1305.1.

Canterbury, G. E., T. E. Martin, D. R. Petit, L. J. Petit, and D. F. Bradford. 2001. Bird Communities and Habitat as Ecological Indicators of Forest Condition in Regional Monitoring. *Conservation Biology* 14(2):544-558. https://doi.org/10.1046/j.1523-1739.2000.98235.x.

Danchin, E., L-A. Giraldeau, T.J. Valone, and R.H. Wagner. 2004. Public Information: From Nosy Neighbors to Cultural Evolution. *Science* 305(5683):487-491. https://doi.org/10.1126/science.1098254.

Darveau, M., P. Beauchesne, L. Bélanger, J. Huot, and P. Larue. 1995. Riparian Forest Strips as Habitat for Breeding Birds in Boreal Forest. *The Journal of Wildlife Management 59*(1):67-78. https://doi.org/10.2307/3809117.

Doligez, B., E. Danchin, and J. Clobert. 2002. Public Information and Breeding Habitat Selection in a Wild Bird Population. *Science* 297(5584):1168-1170. https://doi.org/10.1126/science.1072838.

Donato, D. C., J. L. Campbell, and J. F. Franklin. 2011. Multiple successional pathways and precocity in forest development: can some forests be born complex? *Journal of Vegetation Science* 23(3):576-584. https://doi.org/10.1111/j.1654-1103.2011.01362.x.

Ewers, R. M., S. Thorpe, and R. K. Didham. 2007. Synergistic interactions between edge and area effects in a heavily fragmented landscape. *Ecology* 88(1):96-106. https://doi.org/10.1890/0012-9658(2007)88[96:SIBEAA]2.0.CO;2.

Franklin, J. F., R. J. Mitchell, and B. J. Palik. 2007. Natural Disturbance and Stand Development Principles for Ecological Forestry. *USDA Forest Service – General Technical Report NRS-19*. https://www.fs.usda.gov/nrs/pubs/gtr/gtr_nrs19.pdf

Franklin, J. F., T. A. Spies, R. Van Pelt, A. B. Carey, D. A. Thornburgh, D. R. Berg, D. B. Lindenmayer, M. E. Harmon, W. S. Keeton, D. C. Shaw, K. Bible, and J. Chen. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. *Forest Ecology and Management* 155(1-3):399-423. https://doi.org/10.1016/S0378-1127(01)00575-8.

Greenwood, P. J., and P. H. Harvey. 1982. The Natal and Breeding Dispersal of Birds. *Annual Review of Ecology and Systematics* 13:1-21. https://doi.org/10.1146/annurev.es.13.110182.000245.

Haas, C. A. 1998. Effects of Prior Nesting Success on Site Fidelity and Breeding Dispersal: An Experimental Approach. *The Auk* 115(4):929-936. https://doi.org/10.2307/4089511.

Hagan, J. M., W. M. Vander Haegen, and P. S. McKinley. 1996 The Early Development of Forest Fragmentation Effects on Birds. *Conservation Biology* 10(1):188-202. https://www.jstor.org/stable/2386955.

Harris, S., and M. Betts. 2021. Bird abundance is highly dynamic across succession in early seral tree plantations. *Forest Ecology and Management* 483:118902. https://doi.org/10.1016/j.foreco.2020.118902.

H.J. Andrews Experimental Forest [HJA]. 2020. *Holiday Farm Fire September 2020*. https://andrewsforest.oregonstate.edu/gallery/holiday-farm-fire-september-2020

Hoover, J. P. 2003. Decision rules for site fidelity in a migratory bird, the prothonotary warbler. *Ecology* 84(2):416-430. https://doi.org/10.1890/0012-9658(2003)084[0416:DRFSFI]2.0.CO;2.

Kim, H., B. C. McComb, S. J. K. Frey, D. M. Bell, and M. G. Betts. 2022. Forest microclimate and composition mediate long-term trends of breeding bird populations. *Global Change Biology* 28(21):6180-6193. https://doi.org/10.1111/gcb.16353.

Laurance, S. G. W., P. C. Stouffer, and W. F. Laurance. 2004. Effects of Road Clearings on Movement Patterns of Understory Rainforest Birds in Central Amazonia. *Conservation Biology* 18(4):1099-1109. https://doi.org/10.1111/j.1523-1739.2004.00268.x.

Laurance, W. F. 2000. Do edge effects occur over large spatial scales?. *Trends in Ecology and Evolution* 15(4):134-135. https://doi.org/10.1016/S0169-5347(00)01838-3.

Mack, D. E.. and W. Yong. 2020. Swainsn’s Thrush (*Catharus ustulatus*), version 1.0. In Birds of the World (A.F. Poole and F.B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi-org.oregonstate.idm.oclc.org/10.2173/bow.swathr.01.

MacKenzie, D. I, Nichols, J. D., Royle, J. A., Pollock, K. H., Bailey, L. L., and Hines, J. E. 2017. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence, Second Edition. Academic Press. https://doi.org/10.1016/C2012-0-01164-7.

Marten, K., and P. Marler. 1977. Sound transmission and its significance for animal vocalization. *Behavioral Ecology and Sociobiology* 2:271-290. https://doi.org/10.1007/BF00299740.

McKenzie, D., Z. Gedalof., D. Peterson, P. Mote. 2004. Climatic Change, Wildfire, and Conservation. *Conservation Biology* 18(4):890-902. https://doi.org/10.1111/j.1523-1739.2004.00492.x.

Nagy, R., E. Fusco, B. Bradley, J. T. Abatzoglou, and J. Balch. 2018. Human-Related Ignitions Increase the Number of Large Wildfires across U.S. Ecoregions. *Fire* 1(1). https://doi.org/10.3390/fire1010004.

Niemi, G. J., and M. E. McDonald. 2004. Application of Ecological Indicators. *Annual Review of Ecology, Evolution, and Systematics* 35:89-111. https://doi.org/10.1146/annurev.ecolsys.35.112202.130132.

O’Connell, T. J., L. E. Jackson, R. P. Brooks. 2000. Bird Guilds as Indicators of Ecological Condition in the Central Appalchians. *Ecological Applications* 10(6):1706-1721. https://doi.org/10.1890/1051-0761(2000)010[1706:BGAIOE]2.0.CO;2.

Perry, D.A., P.F. Hessburg, C.N. Skinner, T.A. Spies, S.L. Stephens, A.H. Taylor, J.F. Franklin, B. McComb, and G. Riegel. 2011. The ecology of mixed severity fire regimes in

Washington, Oregon, and Northern California. *Forest Ecology and Management* 262(5):703-717. https://doi.org/10.1016/j.foreco.2011.05.004.

Phalan, B., J. Northrup, Z. Yang, R. Deal, J. Rousseau, T. Spies, and M. Betts. 2019. Impacts of the Northwest Forest Plan on forest composition and bird populations. *PNAS* 116(8):3322-3327. https://doi.org/10.1073/pnas.1813072116.

Potts, J. R., T. Hillen, and M. A. Lewis. 2016. The "edge effect" phenomenon: deriving population abundance patterns from individual animal movement decisions. *Theoretical Ecology* 9:233-247. https://doi.org/10.1007/s12080-015-0283-7.

Simons, T. R., K. H. Pollock, J. M. Wettroth, M. W. Alldredge, K. Pacifici, J. Brewster. 2009. Sources of Measurement Error, Misclassification Error, and Bias in Auditory Avian Point Count Data. In: Thomson, D.L., Cooch, E.G., Conroy, M.J. (eds) Modeling Demographic Processes In Marked Populations. Environmental and Ecological Statistics, vol 3. Springer, Boston, MA. https://doi.org/10.1007/978-0-387-78151-8_10.

Schieck, J. 1997. Biased Detection of Bird Vocalizations Affects Comparisons of Bird Abundance among Forested Habitats. *The Condor* 99(1):179-190. https://doi.org/10.2307/1370236.

Spies, T. A. 1998. Forest structure: a key to the ecosystem. *Northwest Science* 72(2):34-39.

Swanson, M. E., N. M. Studevant, J. L. Campbell, and D. C. Donato. 2014. Biological associates of early-seral pre-forest in the Pacific Northwest. *Forest Ecology and Management* 324:160-171. https://doi.org/10.1016/j.foreco.2014.03.046.

Tepley, A. J., F. J. Swanson, T. A. Spies. 2014. Post-fire tree establishment and early cohort development in conifer forests of the western Cascades of Oregon, USA. *Ecosphere* 5(7) 1-23. https://doi.org/10.1890/ES14-00112.1.

United States Department of Agriculture [USDA]. 2020. *Burned Area Emergency Response Summary–Holiday Farm Fire*. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd839860.pdf

Westerling, A. L., H. G. Hidalgo, D. R. Cayan, T. W. Swetnam. Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science* 313(5789) :40-943. https://doi.org/10.1126/science.1128834.

APPENDIX A

Table 3. Full output from the vegetation selection model with 95% confidence intervals for parameters.

Species	$\hat{\beta}_0$	$\hat{\beta}_0$ CI	$\hat{\beta}_0$ p-value	$\hat{\beta}_1$	$\hat{\beta}_1$ CI	$\hat{\beta}_1$ p-value
BTYW	-1.4286	(-1.8375, -1.0202)	7.26×10^{-12}	0.0664	(-0.4230, 0.5558)	0.0664
CBCH	-0.3083	(-0.4931, -0.1236)	-0.0011	-0.4391	(-0.6733, -0.2049)	0.0002
SWTH	-0.5783	(-0.7720, -0.3847)	4.80×10^{-9}	-0.4797	(-0.7323, -0.2271)	0.0002
PAWR	-0.3537	(-0.5404, -0.1670)	0.0002	-0.1948	(-0.4052, 0.0156)	0.0696
WEFL	-0.6283	(-0.8509, -0.4057)	3.16×10^{-8}	-0.4547	(-0.7154, -0.1941)	0.0006
HEWA	-1.2563	(-1.6016, -0.9110)	9.94×10^{-13}	0.3708	(-0.0020, 0.7435)	0.0512

Species	$\hat{\beta}_2$	$\hat{\beta}_2$ CI	$\hat{\beta}_2$ p-value	$\hat{\beta}_3$	$\hat{\beta}_3$ CI	$\hat{\beta}_3$ p-value	b_i
BTYW	0.3302	(-0.5683, 1.2288)	0.4710	-0.7596	(-1.9925, 0.4733)	0.2270	1.371×10^{-8}
CBCH	0.0640	(-0.4936, 0.6216)	0.8220	-0.3570	(-1.1179, 0.4039)	0.3578	0.0506
SWTH	-0.7199	(-1.3865, -0.0534)	0.0343	0.4797	(-0.3594, 1.3188)	0.2625	0.0381
PAWR	0.3511	(-0.0806, 0.7829)	0.1109	-0.4543	(-0.9839, 0.0754)	0.0928	0.0750
WEFL	0.5227	(0.0189, 1.0264)	0.0420	-0.4499	(-1.0823, 0.1826)	0.1633	0.1443
HEWA	0.3644	(-0.6113, 1.3400)	0.4642	-1.0029	(-2.4107, 0.4049)	0.1626	0.0172

Table 4. Full output from the concussion effect model with 95% confidence intervals for parameters.

Species	$\hat{\beta}_0$	$\hat{\beta}_0$ CI	$\hat{\beta}_0$ p-value	$\hat{\beta}_1$	$\hat{\beta}_1$ CI	$\hat{\beta}_1$ p-value
BTYW	-1.7918	(-2.5324, -1.0511)	2.12×10^{-6}	0.4595	(-0.4068, 1.3259)	0.2990
CBCH	-0.4434	(-0.7162, -0.1706)	0.0015	-0.4389	(-0.8209, -0.0569)	0.0243
SWTH	-0.6228	(-0.9161, -0.3295)	3.16×10^{-5}	-0.6248	(-1.0240, -0.2255)	0.0022
PAWR	-0.3764	(-0.6414, -0.1114)	0.0054	-0.2275	(-0.5302, 0.0752)	0.1408
WEFL	-0.4969	(-0.7927 -0.2012)	0.0010	-0.8823	(-1.2766, -0.4879)	1.16×10^{-5}
HEWA	-1.3097	(-1.8049, -0.8146)	2.17×10^{-7}	0.4291	(-0.1194, 0.9776)	0.1250

Species	$\hat{\beta}_2$	$\hat{\beta}_2$ CI	$\hat{\beta}_2$ p-value	$\hat{\beta}_3$	$\hat{\beta}_3$ CI	$\hat{\beta}_3$ p-value	b_i
BTYW	0.5754	(-0.3128, 1.4635)	0.2040	-0.6221	(-1.673, 0.4287)	0.2460	6.37×10^{-11}
CBCH	0.2804	(-0.0738, 0.6347)	0.1208	-0.0159	(-0.5156, 0.4838)	0.9503	0.0174
SWTH	0.0762	(-0.3052, 0.4577	0.6953	0.2471	(-0.2688, 0.7630)	0.3478	0.0429
PAWR	0.0311	(-0.3426, 0.4048)	0.8705	0.0636	(-0.3579, 0.4851)	0.7675	0.0886
WEFL	-0.2747	(-0.7047, 0.1554)	0.2106	0.7945	(0.2600, 1.3290)	0.0036	0.1349

HEWA	0.1100	(-0.5515, 0.7715)	0.7440	-0.1154	(-0.8682, 0.6374)	0.7640	0.0129