

Burn severity and heterogeneity mediate avian response to wildfire in a hemiboreal forest

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ABSTRACT

Recent studies have shown the importance of accounting for burn severity when assessing the effects of forest fires on avian communities. We add to this growing literature with one of the first studies to assess these effects in boreal and hemiboreal regions of North America. We conducted point counts in control and treatment areas for 2–3 years before (2009–2011) and 4–5 years after (2012–2016) the Pagami Creek Fire in northern Minnesota, USA. Our primary objectives were to (1) assess the effects of burn status, burn severity and burn heterogeneity on avian communities and patterns of abundance in individual bird species and (2) compare these results when counts were either restricted to 100 m count radii or corrected for potential detection heterogeneity. Species richness and total abundance of birds were best described by a burn severity by year interaction, with lower-severity burns having similar patterns to control areas and higher-severity burns showing reduced abundance and richness following the fire. Nevertheless, values associated with higher-severity burns returned to near-control levels after five years. Of 43 species analyzed, 27 showed detectable responses to the fire through shifts in abundance. Nearly equal proportions of these species had generally positive versus negative effects of the fire. Sixteen (59%) of these fire responses were mediated by burn severity, while four (15%) and seven (26%) were best predicted by burn heterogeneity or burn status, respectively. Of the species whose response to the fire was mediated by burn severity, the majority (81%) had decreased abundance as burn severity increased. However, all species responding to burn heterogeneity or burn status showed positive relationships to these metrics. Results were similar between analytical methods that adjusted for species detectability vs those that did not, with 78% of species having the same interpretation of fire effects. Burn severity and heterogeneity, in addition to the sole distinction of whether an area was burned, dictate the response of birds to forest fire in hemiboreal forests. Mixed-severity fires, which include a range of burn severities and associated vegetation change, will likely benefit the most bird species in this region. When possible, management should focus on retaining this spectrum of post-fire conditions, though of primary importance is the inclusion of wildfire in forest planning and conservation efforts.

1. Introduction

A recent paradigm in fire ecology identifies burn severity and burn heterogeneity as dominant drivers mediating the effects of forest fires on birds and bird communities (Fontaine and Kennedy, 2012; Hutto and Patterson, 2016; Taillie et al., 2018; Tingley et al., 2016). Studies have generally found that bird species respond uniquely to burn severity

levels as opposed to simply the presence of fire. Once severity is accounted for, the response of many bird species becomes clear and often a positive relationship with fire is detected (Smucker et al., 2005). This is especially true when post-fire time series are long and the time-since-fire is taken into account (Hutto and Patterson, 2016). Based on these results and those relating to other ecosystem processes, scientists are recommending the use of severe fires and associated mixed-severity

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impacts as a means of maintaining biodiversity (Bond et al., 2012; DellaSala et al., 2017; Hutto et al., 2015; Kelly et al., 2017).

Mixed-severity fires create a patchwork of vegetation, comprising areas that are unburned, lightly burned and areas where complete stand turnover has occurred (DellaSala and Hanson, 2015). This landscape heterogeneity provides habitat for bird species exhibiting a range of responses to fire (Hutto et al., 2015; Hutto and Patterson, 2016). Benefits extend not only to archetypal species like Black-backed Woodpecker (*Picoides arcticus*; Tremblay et al., 2016), but also species that utilize residual live trees and edges like Olive-sided Flycatcher (*Contopus cooperi*; Baker et al., 2016) and species showing delayed responses to post-fire succession such as shrub nesters (Stephens et al., 2015).

To date, studies assessing the effects of burn severity and heterogeneity on birds have primarily been conducted in western forests (Hutto and Patterson, 2016; Kotliar et al., 2007; Smucker et al., 2005; Stephens et al., 2015; Tingley et al., 2016) as well as Appalachian forests (Greenberg et al., 2013; Rose et al., 2016; Rush et al., 2012). Although boreal and hemiboreal forests have received much attention regarding effects of fire on birds (e.g., Haney et al., 2008; Hannon and Drapeau, 2005; Niemi, 1978; Schieck and Song, 2006; Schulte and Niemi, 1998), the effects of burn severity and heterogeneity have rarely been tested across the bird community (but see Azeria et al., 2011; Knaggs, 2018).

In this study, we compare the effects of fire, burn severity and burn heterogeneity on bird species and community patterns before and after the Pagami Creek Fire (2011) in the hemiboreal forests of northern MN, USA. We used a before-after control-impact study design with point counts collected for 2–3 years pre-fire and 4–5 years post-fire at unburned controls and across a continuum of burn severities. We predicted that most individual bird species would show responses to the fire, with this effect generally depending on burn severity or burn heterogeneity, as opposed to solely the presence of fire disturbance. We expected community patterns in the highest burn severities to be most divergent from unburned controls. In addition, we compare the effects of the above parameters when bird data have been corrected with density estimates that account for detectability (e.g., Stephens et al., 2015) versus raw point-count data restricted to 100 m from observers (e.g., Smucker et al., 2005).

2. Methods

2.1. Study area

The study was conducted near the southern edge of the boreal forest in the Superior National Forest (SNF; 48°N 92°W) of northeastern Minnesota (Fig. 1). The SNF is approximately 1.6 million ha and is primarily managed for multiple uses such as timber harvest and recreation. However, it also contains a 440,000 ha wilderness area, the Boundary Waters Canoe Area Wilderness, where motorized travel is limited and timber harvesting is prohibited. Wildfire was historically the dominant disturbance affecting forests in this landscape (Heinselman, 1996; Niemi et al., 2016). Federal land management agencies, including the US Forest Service, can use naturally ignited wildland fires to accomplish specific resource management objectives (e.g., fuel reduction) in designated areas such as wilderness where doing so does not conflict with higher priorities like human safety and property protection (USDA/USDI, 2003). The Pagami Creek Fire (PCF) started by a lightning strike in a remote area of the Boundary Waters Canoe Area Wilderness in late summer 2011 and burned as a low-severity fire until weather conditions changed dramatically in September, converting the fire into wildfire status. The wildfire burned approximately 38,000 ha, dominated by high-severity impacts, but including a range of fire severities, particularly on its southern edge (Fig. 1; Cooley et al., 2016).

2.2. Bird surveys

Point-count locations from two previously initiated avian research projects were affected by the PCF. Two transects of count locations from a study assessing the effects of wilderness versus conventional forest management (Zlonis and Niemi, 2014) were largely burned by the fire. In addition, portions of one transect from a long-term study of forest birds in the National Forests of the Western Great Lakes region (Niemi et al., 2016) was burned by the PCF. Hereafter we refer to the former study as the Wilderness Management Study (WMS) and the latter study as National Forest Bird (NFB). In addition to 31 burned count locations, we retained most unburned control points from within approximately 5 km of the PCF boundary for a total of 83 count locations sampled pre and post-fire (Fig. 1). These points had previously been grouped into seven unique transects of 12 point counts. Each individual transect was surveyed by one observer on one morning. All counts were off-road, situated within the vegetation being sampled to reduce the effects of potential roadside bias (Hanowski and Niemi, 1995). To limit the effects of increased numbers of bird surveyors on detectability, we omitted data from one potential control transect (nine count locations) that would have added six unique bird surveyors over the duration of the pre and post-fire timeline.

Seventy-two count locations in the WMS were surveyed for breeding birds three times in a two-year period before the PCF (2010–2011; Zlonis and Niemi, 2014) and annually for 5 years after the fire (2012–2016). Twelve NFB count locations were sampled annually starting in 1991. However, we restricted data to a similar survey period as the WMS, with three pre-fire samples (2009–2011) and five post-fire samples (2012–2016). Three of the NFB count locations were not sampled in 2011–2012 and thus have two pre-fire samples and four post-fire samples. In addition, data from one of the WMS count locations was censored due to consistent noise interference from a nearby creek. A total of 655 10-minute point counts, conducted at 83 locations, were included in analysis.

We used 10-min, unlimited distance, point counts to collect data on breeding birds (Niemi et al., 2016; Ralph et al., 1995). All surveys were conducted during the peak breeding period of territorial passerines, between 19 May and 25 June, in generally good weather conditions; low winds and little or no precipitation. The specific survey protocols are described in more detail elsewhere (Niemi et al., 2016; Zlonis and Niemi, 2014). Protocols were nearly identical between the WMS and NFB data, with only minor differences in the number and size of distance bins used; NFB identified birds as either within 10 m, 11–25 m, 26–50 m, 51–100 m, or > 100 m from the observer, whereas the WMS used 0–25 m, 26–50 m, 51–100 m and > 100 m bins. Four different observers collected data, three of which collected 98% of counts. All observers were tested for their ability to identify songs and calls of regional birds and a standardized training session was conducted in spring prior to surveys (Zlonis et al., 2016).

2.3. Fire covariates

We determined the burn status of each point by assessing the proportion of post-fire canopy change surrounding each count location. Using the global forest cover change maps developed by Hansen et al. (2013), we summarized the proportion of 30 m grid cells that had been converted from forest to non-forest from pre-fire (2010) to post fire (2011–2014) in 125 m buffers around each point. Open water or other open wetland cover types that could not exhibit canopy change were removed from each buffer prior to the change analysis. Plots were considered burned if $\geq 10\%$ of the buffer experienced canopy change after the fire. Each burned point was visually examined in a GIS and on the ground to ensure that the precursor to canopy change was fire as opposed to other disturbances such as wind, insect damage, beaver (*Castor Canadensis*) activity or forest management. Small canopy disturbances had occurred at many plots, but only burned plots met the

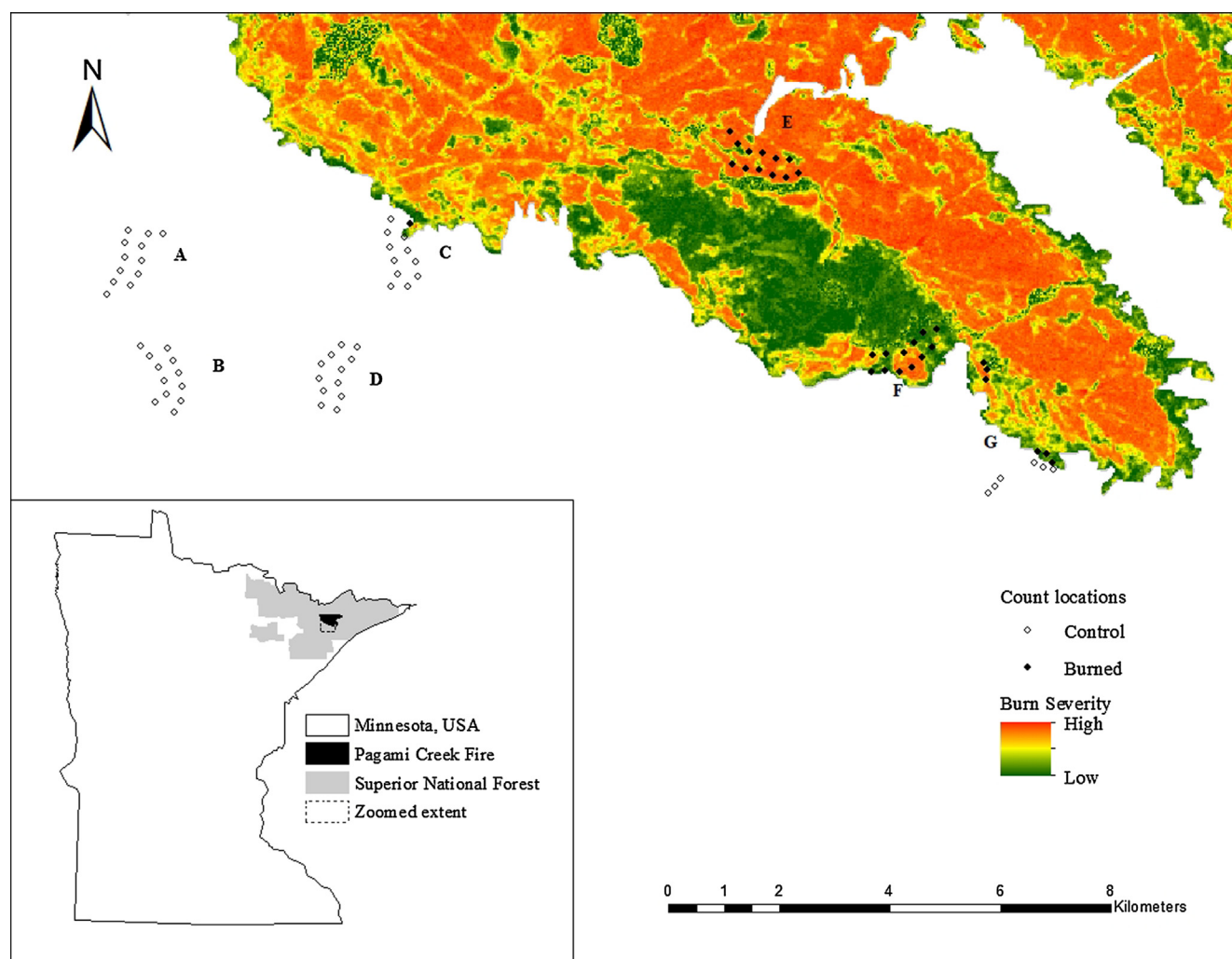


Fig. 1. Pagami Creek Fire burn-area and associated point-count locations within the Superior National Forest of northeastern Minnesota, USA. Point-counts were grouped into seven transects, which were sampled by one observer on a given morning from 2009 to 2016. Transects A, B and D consisted only of control points. Transects C and G had both control and burned points. Transects E and F consisted only of burned points.

10% minimum criteria.

Burn severity was characterized in each buffer using the relative difference normalized burn ratio (RdNBR; Miller and Thode, 2007). A raster map of RdNBR across the PCF was developed using Landsat-5 sensor images from June 26, 2009 (pre-fire) and October 6, 2011 (post-fire), see Cooley et al. (2016) for detailed methodology. The severity for each burned point-count location was calculated by averaging RdNBR values across the grid cells in each 125 m buffer. Ground-based estimates of tree crown scorch (Jain and Graham, 2007) and RdNBR values were highly correlated in the PCF ($R^2 = 0.82$, P. Wolter unpublished data). The most severely burned sites had all trees killed by the fire and were usually barren of live vegetation immediately post-fire, whereas low-severity burns created a patchwork of vegetation mortality in both the over and under-story but left most trees alive. Several studies have utilized RdNBR and its antecedent, normalized burn ratio (Key and Benson, 2005), to assess the effects of burn severity on birds (Kotliar et al., 2007; Rose et al., 2016; Tingley et al., 2016).

Burn heterogeneity, sometimes referred to as burn diversity, was calculated as the standard deviation of RdNBR burn severity estimates across grid cells within 125 m circle surrounding each point-count location (after Azeria et al., 2011; Tingley et al., 2016). We primarily use the term heterogeneity in this context, as diversity implies a broader ecological scale that disregards the spatial pattern of variation. At burned locations, there was no statistical relationship between burn

severity and burn heterogeneity ($R^2 = 0.01$). The burn severity metric inherently masks variations in severity by averaging across the count circle and does not necessarily represent a homogenous value. Conversely, burn heterogeneity accounts for this variation, with high values representing count circles that include both severely burned and unburned forest patches and low values representing more homogenous burns of a specific severity level. Values of burn severity and heterogeneity at bird count locations were representative of the broader PCF (Table 1). Unburned control points were assigned zero values for both metrics.

Table 1

Comparison of burn severity and heterogeneity values at bird count locations and across the entire Pagami Creek Fire. Burn severity represents the 5th percentile, 95th percentile and mean RdNBR values within 125 m of point-count locations. Burn heterogeneity represents the same metrics, but for the standard deviation of RdNBR values within 125 m of point-count locations. Values for the entire fire were calculated in a moving window analysis across potential 125 m circles within the fire boundary.

Fire covariate	5%	95%	Mean
Burn severity, entire fire	8.34	44.57	33.71
Burn severity, bird count locations	5.13	42.43	25.58
Burn heterogeneity, entire fire	1.37	12.68	4.98
Burn heterogeneity, bird count locations	1.71	13.8	6.9

2.4. Statistical analyses

2.4.1. Bird response to fire

Bird species were split into two analysis groups depending on their life histories and numbers of detections. Group One included 40 territorial passerine species, each with at least 20 observations of singing males (Appendix A). These species were included in an analysis comparing results when counts were either corrected for potential detection heterogeneity using a modelling approach or not. For analyses of uncorrected counts, we reduced the effects of possible detection heterogeneity by utilizing only observations from within 100 m of observers, which (1) limited potential distance sampling issues where birds in open and burned areas could be detected from great distances and (2) ensured birds were utilizing vegetation from the same area that fire covariate information was derived. Additionally, we included only singing observations, which were entirely detected aurally, to limit any potential effects of increased visual observations in open and burned landscapes. Counts were corrected for detectability using a combination of removal and distance models implemented in the QPAD framework (Sólymos et al., 2013), described in detail in Section 2.4.2.

Group Two consisted of eight species whose vocalizations are not easily categorized as territorial or distinguishable between sexes, but had at least 20 non-flying observations within 100 m of point-count observers (Appendix A). These species are largely in families Picidae, Paridae, Sittidae and Corvidae. We did not compare models developed from corrected and uncorrected counts in these species, as estimates of detectability are most reliable when only data from singing males are used (Buckland et al., 2008). In addition to individual bird species, we also developed two community-based response variables, species richness and total bird abundance. Both were summarized at the point-count level and included all non-flyover observations within 100 m of observers. All species included in analyses are considered regular breeding species in the SNF.

To test the effects of burn status, burn severity and burn heterogeneity on birds, we developed eight candidate models for each individual bird species as well as community metrics of total bird abundance and species richness (Table 2). We utilized a similar before-after control-impact framework and modelling strategy as Stephens et al. (2015), where temporal (year) and fire-related effects (status, severity, heterogeneity) were included in both additive and interactive models. For species where the interactive model was most supported, we concluded that the fire had caused the observed variation in bird abundance. For species where an additive model was most supported (i.e., effect of fire did not vary across years), we visually inspected patterns of detections before and after the fire to determine if the fire had influenced the observed result (after Stephens et al., 2015). For species where either the temporal or null model were most supported, we concluded no significant fire effects.

For each response variable, we developed and compared generalized linear mixed models (GLMM) for the eight candidate models listed in Table 2. Response variables were either point-count abundance of

individual bird species (48 total species; Appendix A) or community summaries of point-count level species richness and total bird abundance. Transect was included as a random effect. We first attempted GLMMs with a Poisson error structure. However, if overdispersion was an issue in one or more candidate models for a given response variable, we reran the analysis with point-count location nested in transect to account for the additional variation. Analyses were conducted in R (R Core Team, 2017) using package lme4 (Bates et al., 2014).

Models were compared and ranked using AICc (Burnham and Anderson, 2002). All models within $\Delta 2\text{AICc}$ of the top model were considered competing models. If there was more than one competing model for a given comparison, the most parsimonious model was selected as the best supported model. For example, if two competing models included the null model and the severity interaction model, the null model would be reported as the best supported model (Table 2).

2.4.2. Accounting for detectability

The above analyses were also completed using count data corrected for detection heterogeneity, which allowed us to compare results between corrected and uncorrected counts. For each Group One species, we employed the QPAD approach developed by Sólymos et al. (2013) to create detectability offsets that were then used to model density (males/ha) in the GLMM framework described above. The method is termed ‘QPAD’ in reference to the four values that can be calculated in the equation:

$$E[Y] = Np(t_j)q(r_k) = DAp(t_j)q(r_k) \quad (1)$$

where Y is the expected count for a given species at a given count location, N is the total abundance, D is the density per unit area, A is the area sampled, $p(t_j)$ is the probability that an individual bird will vocalize at least once during the count interval with total time t_j , and $q(r_k)$ is the probability that a given individual will be detected given it is vocalizing within the point-count radius r_k (Sólymos et al., 2013). The basic components of detectability functions, p and q , are estimated independently using removal (p) and distance (q) sampling methods, while the estimated density, D , is calculated using combined estimates of p , q , and A . Each component is determined using a conditional maximum likelihood approach in the R package ‘detect’ (Sólymos et al., 2013; Solymos et al., 2018). The method allows for inclusion and assessment of site and survey level covariates hypothesized to affect detectability (e.g., tree cover) in both distance and removal models and can accommodate varying survey protocols. In our case, we leveraged a large amount of point-count data (> 12,000 counts) to parameterize detectability offsets for each species. These data were collected across Minnesota during five research projects including WMS and NFB and initially analyzed as part of the Minnesota Breeding Bird Atlas (Pfannmuller et al., 2017).

We developed nine removal and three distance models based on combinations of covariates tested in Sólymos et al. (2013). Site-level covariates included the openness of vegetation at the point count, ‘LCC’, and the proportion of tree cover, ‘TREE’. Survey-level covariates

Table 2

List of statistical models compared for individual bird species and community metrics. Eight candidate generalized linear mixed models were ranked using an AIC framework. The interpretation of fire effects is indicated when a given model was selected as the top model (after Stephens et al., 2015). Abbreviations and simplified model names are provided.

Model	Interpretation	Abbreviation
$y = 1 + (1 \mid \text{transect})$	No fire effect	N; “Null”
$y = \text{year} + (1 \mid \text{transect})$	No fire effect	Y; “Year”
$y = \text{year} + \text{burn} + (1 \mid \text{transect})$	Potential effect of burn status	B; “Burned additive”
$y = \text{year} + \text{burn} + \text{burn}*\text{year} + (1 \mid \text{transect})$	Effect of burn status	B [*] ; “Burned interaction”
$y = \text{year} + \text{severity} + (1 \mid \text{transect})$	Potential effect of burn severity	S; “Severity additive”
$y = \text{year} + \text{severity} + \text{severity}*\text{year} + (1 \mid \text{transect})$	Effect of burn severity	S [*] ; “Severity interaction”
$y = \text{year} + \text{heterogeneity} + (1 \mid \text{transect})$	Potential effect of burn heterogeneity	H; “Heterogeneity additive”
$y = \text{year} + \text{heterogeneity} + \text{heterogeneity}*\text{year} + (1 \mid \text{transect})$	Effect of burn heterogeneity	H [*] ; “Heterogeneity interaction”

Table 3

Results of generalized linear mixed models and AIC-based model selection for individual bird species and community metrics. Competing models (ΔAIC_c) are listed for each species and metric. The top model selected by parsimony is bolded and the model with lowest AICc is italicized. When applicable, results are reported for both abundance models and detectability-corrected QPAD models. The overall interpretation of fire effects (positive; (+) or negative; (–)) as well as the specific fire covariate from the top model are reported. Interpretations are compared between analysis methods. Model abbreviations can be found in Table 2.

Common name	Analysis Group	No Offset		QPAD Offset		Comparison
		Competing models	Interpretation	Competing models	Interpretation	
Species richness		<i>S</i> [*]	(–) burn severity			
Abundance		<i>S</i> [*]	(–) burn severity			
Alder Flycatcher	Group1	H , <i>H</i> [†]	(+) burn heterogeneity	S , <i>S</i> [*]	(+) burn severity	Different [*]
American Redstart	Group1	N , Y, B, H	No fire effect	N , Y, B, S, H [*]	No fire effect	Same
American Robin	Group1	B , <i>B</i> [*]	(+) burn status	S , <i>S</i> [*]	(+) burn severity	Different [*]
Black-and-white Warbler	Group1	S	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Blackburnian Warbler	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Black-throated Green Warbler	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Blue-headed Vireo	Group1	B , S, <i>S</i> [*]	No fire effect [†]	S , <i>S</i> [*] , <i>B</i> [*]	No fire effect [†]	Same
Brown Creeper	Group1	N , Y, B, H, <i>B</i> [*]	No fire effect	N , B, H, S, <i>S</i> [*] , <i>B</i> [*]	No fire effect	Same
Canada Warbler	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Cape May Warbler	Group1	S	No fire effect [†]	–	–	–
Chestnut-sided Warbler	Group1	B[*]	(+) burn status	B[*]	(+) burn status	Same
Chipping Sparrow	Group1	H	(+) burn heterogeneity	S , <i>S</i> [*]	(+) burn severity	Different [*]
Common Yellowthroat	Group1	S[*], <i>H</i>[†], <i>B</i>[*]	(+) burn severity	S[*], <i>B</i>[*]	(+) burn severity	Same
Connecticut Warbler	Group1	Y , S, B, H	No fire effect	S , H, <i>S</i> [*]	No fire effect [†]	Same
Dark-eyed Junco	Group1	Y , S, B, H, <i>S</i> [*] , <i>B</i> [*]	No fire effect	S[*]	(+) burn severity	Different
Golden-crowned Kinglet	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Hermit Thrush	Group1	N , Y, <i>B</i> [*]	No fire effect	Y , S, B, H, <i>S</i> [*]	No fire effect	Same
House Wren	Group1	S , <i>S</i> [*]	(+) burn severity	S , <i>S</i> [*]	(+) burn severity	Same
Least Flycatcher	Group1	Y , S, B, H	No fire effect	–	–	–
Lincoln's Sparrow	Group1	H , <i>H</i> [†]	(+) burn heterogeneity	H , <i>H</i> [†] , <i>S</i> [*]	(+) burn heterogeneity	Same
Magnolia Warbler	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Mourning Warbler	Group1	B , <i>B</i> [*]	(+) burn status	–	–	–
Yellow-rumped Warbler	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Nashville Warbler	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Northern Waterthrush	Group1	B , <i>B</i> [*]	(+) burn status	B , <i>B</i> [*]	(+) burn status	Same
Olive-sided Flycatcher	Group1	N , Y, H, S, <i>B</i> [*] , <i>H</i> [†] , <i>S</i> [*]	No fire effect	S , <i>B</i> [*] , <i>S</i> [*]	(+) burn severity	Different
Ovenbird	Group1	S , <i>S</i> [*]	(–) burn severity	S[*]	(–) burn severity	Same
Purple Finch	Group1	N , S, <i>S</i> [*]	No fire effect	–	–	–
Red-eyed Vireo	Group1	S[*]	(–) burn severity	Y , H, S, B, H [*]	No fire effect	Different
Ruby-crowned Kinglet	Group1	S , <i>S</i> [*]	No fire effect [†]	N , S, <i>S</i> [*] , <i>B</i> [*]	No fire effect	Same
Song Sparrow	Group1	S[*]	(+) burn severity	S[*]	(+) burn severity	Same
Swamp Sparrow	Group1	H , <i>H</i> [†]	(+) burn heterogeneity	S[*]	(+) burn severity	Different [*]
Veery	Group1	N , Y	No fire effect	N	No fire effect	Same
White-throated Sparrow	Group1	B , <i>B</i> [*]	(+) burn status	B , <i>B</i> [*]	(+) burn status	Same
Winter Wren	Group1	S , <i>S</i> [*]	(–) burn severity	S , <i>S</i> [*]	(–) burn severity	Same
Yellow-bellied Flycatcher	Group1	S[*]	(–) burn severity	S[*]	(–) burn severity	Same
Black-backed Woodpecker	Group2	B , <i>B</i> [*]	(+) burn status	–	–	–
Black-capped Chickadee	Group2	S , B, H	No fire effect [†]	–	–	–
Blue Jay	Group2	N , Y	No fire effect	–	–	–
Cedar Waxwing	Group2	B[*]	(+) burn status	–	–	–
Canada Jay	Group2	N , Y, S	No fire effect	–	–	–
Northern Flicker	Group2	N , Y, S, H, B, <i>S</i> [*] , <i>B</i> [*]	No fire effect	–	–	–
Red-breasted Nuthatch	Group2	S , <i>S</i> [*]	(–) burn severity	–	–	–

[†] Species with a top-selected additive model that was not interpreted, see text (after Stephens et al., 2015).

^{*} Same overall fire effect (positive or negative) between analysis methods.

included the time of day of the count (time since local sunrise), time of year of the count (Julian day) and quadratic transformations of these variables. We ran each model for all Group One species and compared with Schwarz's Bayesian information criterion (BIC; Schwarz, 1978). The removal and distance models with the lowest BIC were selected as the best models and used to calculate density offsets for each point count, which were then applied in the GLMM process described in Section 2.4.1. For each species, we compared the results of GLMM analyses using counts adjusted with density offsets to those using unadjusted, 100 m radius, counts. However, we use the latter results when reporting the principal findings in Section 3.2 below.

3. Results

3.1. Fire effects on bird community

In total, we had 11,284 bird detections of 98 species on 655 counts

from 2009 to 2016 (Appendix A). On a per-point-count level, both species richness and total abundance of birds were best predicted by the severity interaction model (*S*^{*}; Table 3). The variation in the numbers of species and individual birds detected at counts varied by burn severity, with lower severity locations having similar counts to controls and higher severity locations showing lower richness and abundances (Fig. 2). This discrepancy was most pronounced in the year immediately following the fire, with counts becoming more similar between control and burned locations by the fifth post-fire sample.

3.2. Fire effects on bird species

We analyzed the effects of fire on 48 species (Appendix A; Table 3). However, due to issues of convergence and overdispersion, we do not present results for Boreal Chickadee, Northern Parula, Rose-breasted Grosbeak, Swainson's Thrush and Tennessee Warbler. Of the remaining 43 species, we determined that 27 (63%) responded to the fire with

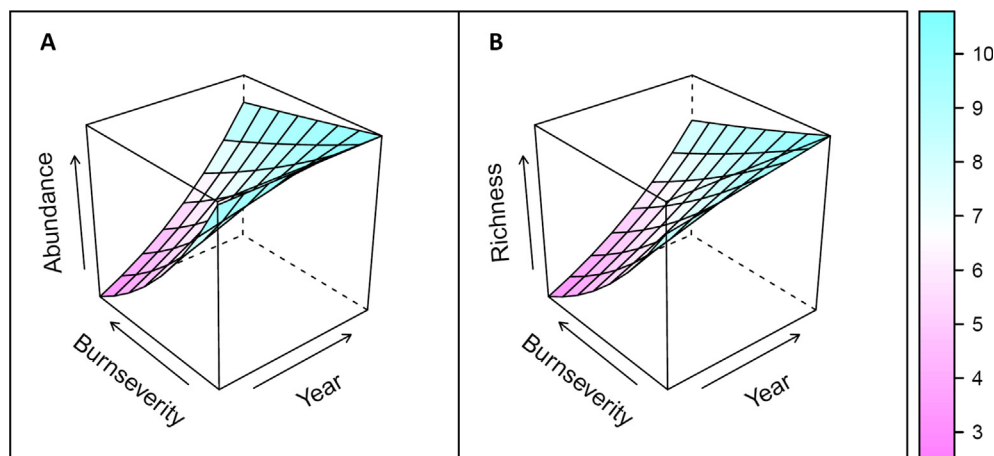


Fig. 2. Model-based predictions of point-count level abundance (A) and species richness (B) depending on year and burn severity. Graphs display predictions from the severity interaction model, which was the best supported model for both metrics. All non-flyover observations from within 100 m of observers were included in the analyses.

detectable shifts in abundance (Table 3). Nearly equal proportions of these species showed generally positive versus negative effects of the fire; when compared to control sites, 14 species exhibited increased abundance at burned sites.

Nineteen species (44%) had the severity-additive (S) or severity-interaction (S^*) model as the top model (Table 3). The severity-interaction model was best supported for four of these species. After examining abundance before and after the fire, we concluded that 12 of the 15 species with the severity-additive model as the top model exhibited clear responses to the fire. Four species (9%) had the heterogeneity-additive model as the top model (H; Table 3). In all of these species, our investigation of pre- and post-fire patterns indicated that fire had caused the observed response to burn heterogeneity. Eight species (19%) responded to burn status, two of which had the burned-interaction model as most supported (B^* ; Table 3). For five of six species with the burned-additive model (B) as best supported, we concluded that the fire had influenced the observed response in predicted abundance. The null (N) and year (Y) models were most supported for nine (21%) and three (7%) species, respectively (Table 3).

Thirteen of 16 species whose response to the fire was mediated by burn severity had decreased abundance as burn severity increased. Example species of this category included Magnolia Warbler and Ovenbird (Fig. 3). The remaining species, Common Yellowthroat, House Wren and Song Sparrow showed higher abundances in high-severity burns (Fig. 3). Though the majority of species showed a positive effect of low-severity fire when compared to high-severity, none of these species had heightened abundance at low-severity sites when compared to unburned controls.

Each of the four species that responded to burn heterogeneity showed increasing abundance with increasing burn heterogeneity; Alder Flycatcher, Chipping Sparrow, Lincoln's Sparrow and Swamp Sparrow. These species had higher abundance in burned areas than controls (Fig. 3).

All species that responded solely to the presence of fire, regardless of severity or heterogeneity, had higher abundance in burned areas over controls. These species included American Robin, Black-backed Woodpecker, Cedar Waxwing, Chestnut-sided Warbler, Mourning Warbler, Northern Waterthrush and White-throated Sparrow. Of these, only the White-throated Sparrow showed an immediate, positive, response to the fire that then decreased with time-since fire (Fig. 3). The remaining species all showed positive effects of the fire that increased with time-since-fire (Fig. 3).

3.3. Detectability comparison

We compared the above results to those developed from QPAD detectability offsets for 32 of the 48 species modelled (Table 3). In addition to the five species which we were not able to accurately model

with abundance data, we were not able to develop useful density offsets for Cape May Warbler, Least Flycatcher, Mourning Warbler and Purple Finch. Twenty-five of 32 species (78%) had the same interpretation of fire effects between abundance and QPAD models (Table 3). Twenty of these species had the identical top model between analyses. The remaining seven species had differing results between the analyses; Alder Flycatcher, American Robin, Chipping Sparrow, Dark-eyed Junco, Olive-sided Flycatcher, Swamp Sparrow and Red-eyed Vireo. However, four of these species had the same general interpretation of fire effects and only varied on the specific fire covariate included in the top model (e.g., severity versus heterogeneity). For the remaining three species that differed in their interpretation between analyses, Dark-eyed Junco, Olive-sided Flycatcher and Red-eyed Vireo, one method concluded no fire effect while the other method identified an effect of the fire.

4. Discussion

Wildfire has long been known to shape the ecological communities of boreal and hemiboreal forests (Heinselman, 1973; Pastor et al., 1996). Yet, the positive effects of mixed-severity fires on biodiversity, and especially avian communities, have only more recently come into focus (Hutto et al., 2015). The varying and positive effects of mixed-severity fire on bird communities have been observed in a variety of forested regions including the Pacific Northwest and Interior west (Smucker et al., 2005; Taillie et al., 2018), Appalachia (Rose and Simons, 2016; Rush et al., 2012), northern boreal forest (Knaggs, 2018) and, as our study demonstrates, the hemiboreal forests of northern Minnesota.

4.1. Avian-community patterns

The abundance and richness of bird communities was most similar between sites experiencing low-severity burns and unburned controls, when compared with sites with high-severity burns. Hence, accounting for the severity of the burn was important beyond the basic designation of whether a site was burned or unburned. Residual vegetation including trees, shrubs and patches of forest were common in lower-severity burns, which provided habitat for many forest-dwelling species common in control sites. These sites also provided habitat for species that responded positively to the presence of fire, regardless of burn severity. Whereas, the complete stand-turnover at high-severity sites, where most or all vegetation was killed, developed conditions suitable for a unique subset of species. This gradient of burn severities embedded in an unburned landscape produces a heterogeneous complex of structure and vegetation communities which supports diverse bird species (Hutto and Patterson, 2016) and the broader ecological processes within forested systems (DellaSala et al., 2017).

At the scale of individual point counts, burn heterogeneity was not

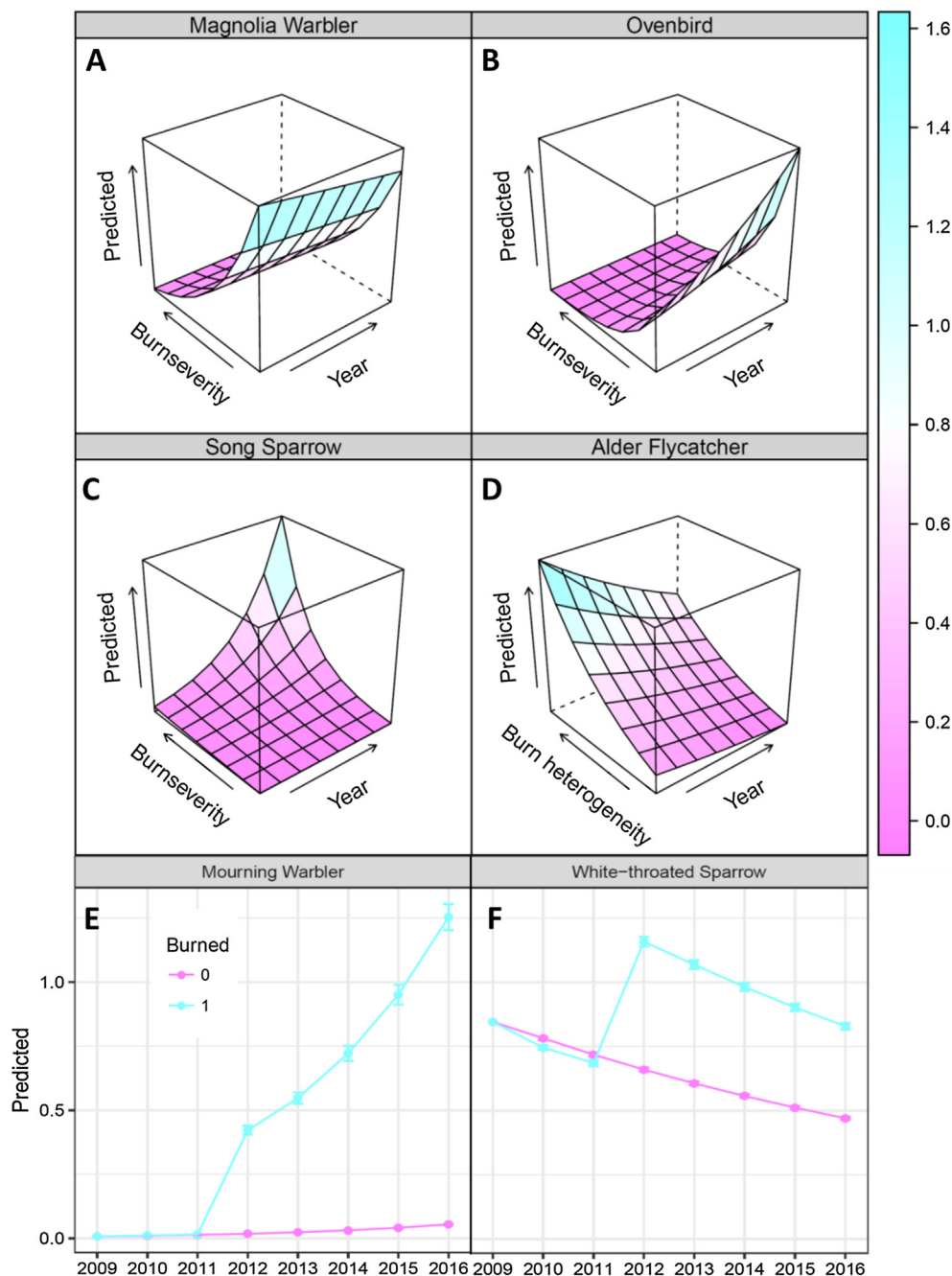


Fig. 3. Model-based predictions of point-count level abundance for selected bird species. Magnolia Warbler, Song Sparrow and Ovenbird (A-C) represent species which responded to burn severity. Alder Flycatcher (D) is an example response to burn heterogeneity. Mourning Warbler and White-throated Sparrow (E-F) were best described by burn-status alone (0 = control, 1 = burned). Sites were burned after sampling in 2011. Error bars on E-F represent 95% confidence intervals around mean predicted counts. Observations of singing males within 100 m of observers were included in analyses.

supported for predicting species richness of birds, contrasting with the findings of Tingley et al. (2016). However, in their study, increasing burn diversity, calculated the same way as heterogeneity, showed the strongest relationship to avian richness when post-fire time-series were long (e.g., 10 years) and the spatial scale of analysis was broad (e.g., across multiple fires). The present study focused on narrower temporal and spatial scales, comparing fire effects at individual point counts within a single fire. Nevertheless, our results support the increasing number of studies that show mixed-severity fires (i.e., diverse fires), benefit the most bird species through producing a range of conditions used by species with various and opposing responses to fire (Taillie et al., 2018 and references therein).

4.2. Species-level patterns

Of the species with clear responses to the fire, slightly more than half responded positively to the fire. These species fell into three groups; seven species increasing in burned areas, three species increasing in higher-severity burns and four species increasing in more heterogeneous burns. In contrast, for all of the 13 species that showed generally negative effects of the fire, the response was mediated by burn severity. Within burned areas, these species were more common in low-severity burns and often not found in high-severity burns.

We interpret these patterns as logical responses by species which evolved with fire disturbance. Species not able to tolerate low-severity

fire or other non-stand replacing disturbances might only find suitable habitat in limited vegetative conditions that would likely be uncommon in many landscapes with natural disturbance regimes. Alternatively, species responding positively to fire, especially high-severity fire, are taking advantage of newly created vegetative conditions and associated habitat features which many species cannot utilize. As succession proceeds, many of these species will decline in abundance and different species will begin to find suitable habitat. This dynamic nature of spatio-temporal landscape patterns of forest cover, structure and age-class distributions produces habitat for the breadth of the avian community in fire-prone forests (Hutto and Patterson, 2016).

Species such as Black-backed Woodpecker and House Wren, which were rarely, or never in the case of House Wren, observed at control sites are likely reliant on recent post-fire conditions to maintain population sources in this region (Niemi, 1978). Neither species is commonly found after logging (Schulte and Niemi, 1998). An additional species which probably falls into this category, but for which we did not sufficiently sample to analyze, is Eastern Bluebird. We detected seven territorial males throughout the study, all in high-severity burns, a pattern observed in other areas of its range (Rose and Simons, 2016). These burn specialists utilize the open structure and high density of snags for foraging and nesting (Azeria et al., 2011; Koivula and Schmiegelow, 2007; White et al., 2016).

The remaining species responding positively to fire were relatively common at unburned control points, indicating potentially less reliance on post-fire conditions for population maintenance. These species are generally categorized as being related to open, wetland and early-successional vegetation; communities which occur in the absence of fire in wetlands and adjacent areas as well as forests affected by logging and natural disturbances like wind-throw. In five years of post-fire sampling, most of these species continued to increase in abundance or stabilized. As was commonly found in other studies, species associated with shrubby and young forests increased dramatically (e.g., Stephens et al., 2015). In the same region as our study, Schulte and Niemi (1998) found that some of these species, namely Common Yellowthroat, Song Sparrow and Lincoln's Sparrow are most associated with post-fire vegetation structure while species like Chestnut-sided Warbler and Mourning Warbler were responding to more general early-successional structure that can develop after logging. Our results suggest that fire disturbance has unique characteristics – distinct from harvest disturbance – to which bird communities respond. These likely include differences in tree and shrub-species composition, densities of live and dead trees and associated structural attributes (Schulte and Niemi, 1998).

Most species-level responses to the fire were mediated by burn severity (59% of fire responses, 37% of all species analyzed). Though slightly different comparisons were made, this figure is intermediate for studies comparing the effects of burn severity to burn status (19–60% of species analyzed; Hutto and Patterson, 2016; Kotliar et al., 2007; Smucker et al., 2005; Stephens et al., 2015). For species that overlapped between studies, we generally found a similar direction and magnitude of response to burn severity. For example, nearly all studies we reviewed found Ovenbird, Golden-crowned Kinglet and Yellow-rumped Warbler decreased with increasing burn severity while House Wren increased with increasing burn severity (Azeria et al., 2011; Fontaine and Kennedy, 2012; Hutto and Patterson, 2016; Knaggs, 2018; Kotliar et al., 2007; Rose and Simons, 2016; Rush et al., 2012; Smucker et al., 2005; Stephens et al., 2015; Taillie et al., 2018), reflecting possible consistency in response to burn severity and associated changes to forest structure across the breeding ranges of many species.

However, our results related to burn severity differed from many studies because we found no clear instances where a species increased in abundance in low-moderate severity burns, but not high severity burns (e.g., Hutto and Patterson, 2016). We think this trend is associated with the heterogeneous nature of forests in this region, where a multitude of disturbances besides fire create vertical and horizontal

structure similar to what develops after low-severity fire. In effect, the post-fire vegetation and structure are not entirely unique in these situations, especially when compared to that of high-severity fire. We did not systematically sample vegetative conditions, which would have allowed us to test this prediction. Ongoing advancements in remote-sensing technologies such as LiDAR will aid in the understanding of forest-structural changes caused by fire.

The responses of four species to fire were best described by increased burn heterogeneity. Three of these species, Alder Flycatcher, Lincoln's Sparrow and Swamp Sparrow, are associated with wetland habitats, ranging from open-water interfaces to shrubby and grassy areas as well as peatlands. Points with high burn heterogeneity most commonly included severe fire with abrupt edges at unburned wetland interfaces, which helps describe their response to this metric. However, the presence of the wetland was not the sole factor driving these relationships, as our before-after sampling and analysis clearly demonstrated increased abundance after the fire at these locations. Rather, it appears to be a synergy of appropriate wetland habitat and an additional factor related to the fire, possibly increased open and shrubby conditions and reduced competition with fewer bird species and individuals utilizing the adjacent severely burned areas. Azeria et al. (2011) found only Black-backed Woodpecker responded positively to burn heterogeneity at the local scale. Though, their analysis included a metric for aquatic edges occurring in severely burned areas, which positively influenced occupancy for wetland-dependent species. The final species showing a positive effect of burn heterogeneity, Chipping Sparrow, is a ground forager common in heavily disturbed areas, but which also nests in trees and shrubs.

Over one third of species analyzed did not have detectable shifts in abundance in relation to the fire. For some of these species, the number of detections within 100 m of count locations was small and affected our power to detect a fire response. For example, the Olive-sided Flycatcher has been shown in a variety of studies to respond positively to fire disturbance (Altman and Sallabanks, 2000) and potentially had a similar response in our study (see QPAD results, Table 3). However, only 20% of the detections for this species were within 100 m and could reliably be used in abundance modelling. Distant detections, though suspected to be related to the fire, were in unknown conditions. For other species, equivocal fire responses might be related to contrasting habitat needs satisfied in both burned and unburned areas. For these species the scale of analysis (e.g., Azeria et al., 2011) and timing of surveys might be especially important. Our methodology was primarily targeted at territorial passerines during the breeding season. Species like Canada Jay had completed breeding by the time of our surveys and likely were transitioning their association with specific vegetation types. Finally, for a subset of species, the transition from burned to unburned conditions might have done little to change their preference for a given area. For example, Brown Creeper, which responds strongly to old-growth trees and snags for nesting and foraging, might have found similar attributes available in pre and post-fire conditions. Much of the forests sampled were old and structurally diverse prior to fire, optimal conditions for this species (Zlonis and Niemi, 2014), while the post-fire conditions might have been similarly beneficial due to increased snag densities (Schulte and Niemi, 1998; White et al., 2016).

One of the primary limitations of our study was the relatively small sample size of burned locations from one fire. This is a manifestation of the before-after nature of the sampling design, where we had no control over the plots affected by fire. As a result, the number of species that we were able to effectively analyze with GLMMs as well as QPAD offsets was reduced. In addition, relatively few species that appeared to respond to the fire had an interaction model as the best supported model, similar to the findings of Stephens et al. (2015). We believe this trend is related to two issues, both affected by sample size constraints; (1) post-fire trends in species' responses (2) the standard we set for our AIC-based model selection. Species with clear responses to the fire that changed directionally after the fire were more likely to have an

interactive model as top model (e.g., Song Sparrow, Fig. 3), whereas species that responded to the fire, but had stable patterns of post-fire response were likely to have the additive model as top model (e.g., Magnolia Warbler, Fig. 3). Both are responses to the fire, but, statistically, the former is possibly easier to detect with smaller sample sizes. Moreover, by selecting the most parsimonious model of competing models as the top model, seven species which had an interactive model with the lowest AICc score instead had additive models selected as the top model. This standard was useful when there were many competing models, but we feel was overly stringent when there were only two, the additive and interactive models of a given fire effect, as was often the case.

4.3. Detectability comparison

Most species we compared had the same interpretation of results between models built with detectability offsets versus unadjusted limited-radius (100 m) counts. In addition, for species that differed, the overall effect of the fire (e.g., positive) was usually the same between modelling strategies and there were no species that had completely contradictory results between analyses, for example, showing a positive effect of burn status in one analysis and a negative effect in the other. The primary difference in findings when using adjusted counts was a higher proportion of species that responded to burn severity as opposed to burn heterogeneity or burn status.

Complex detectability issues related to singing biology likely affected the differing conclusions of fire effects between analyses for Dark-eyed Junco, Olive-sided Flycatcher, and Red-eyed Vireo. For example, both Dark-eyed Junco and Red-eyed Vireo can have very high abundance at individual count locations and their songs are known to be confused with other species e.g., Pine Warbler and Chipping Sparrow for the former or Blue-headed Vireo and Philadelphia Vireo for the latter. Olive-sided Flycatcher have large territories and can be heard singing from great distances, making distance estimation challenging.

We chose to interpret the findings from unadjusted counts for three reasons; (1) these data were truncated to records within 100 m to ensure that birds were utilizing the local vegetative conditions and, thus, the fire covariates we derived (e.g., Hutto, 2016), (2) we were able to analyze additional species that were detected with non-territorial observations and (3) results from the detectability comparison did not highlight major differences between methodologies. Our results indicate that accounting for detectability, though a perennial consideration when using point-count data, can lead to similar conclusions as analyses using unadjusted counts. Using data from the NFB project, Etterson et al. (2009) found a similar pattern and concluded that

methodological and sampling design considerations, many of which overlap with this study, reduced potential detection heterogeneity. However, we encourage further comparisons of statistical methods and detectability corrections, especially in studies similar to ours where dramatic vegetation change and associated changes in aural conditions might confound data collected using point counts (Yip et al., 2017).

5. Conclusions

We found about two thirds of bird species analyzed responded to the fire with changes in abundance. Half of these increased in burned areas, which is similar to many forested regions of North America (Kotliar et al., 2007; Smucker et al., 2005; Stephens et al., 2015; Taillie et al., 2018), though somewhat lower than others (Hutto and Patterson, 2016; Knaggs, 2018; Rose and Simons, 2016). Likewise, we also found that many of the species-specific and bird-community responses to the fire were mediated by burn severity and, to a lesser degree, burn heterogeneity. Historically, hemiboreal forests of northern Minnesota experienced a mix of fire severities, depending on landform type, with fire-return intervals averaging around 50–100 years (Heinselman, 1973; USGS, 2010). Though our findings suggest that bird species in this region are adapted to these conditions, fire regimes have already deviated substantially since settlement through suppression (Frelich and Reich, 1995) and likely will continue to change, trending towards more frequent and severe fires (Sturtevant et al., 2012; Weber and Flannigan, 1997). Ongoing shifts could perhaps be moderated by human influence (Campos-Ruiz et al., 2018). Our results suggest that incorporating wildfire and, ideally, a range of fire severities into forest management will best accommodate the needs of the diverse avifauna in hemiboreal regions.

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Appendix A

List of bird species observed at point counts before and after the Pagami Creek Fire. The Analysis Group and total count of individuals are indicated. All point-count observations are included in the count.

Common name	Scientific name	Analysis	Count
Alder Flycatcher	<i>Empidonax alnorum</i>	Group1	223
American Bittern	<i>Botaurus lentiginosus</i>		3
American Crow	<i>Corvus brachyrhynchos</i>		23
American Goldfinch	<i>Spinus tristis</i>		38
American Kestrel	<i>Falco sparverius</i>		1
American Redstart	<i>Setophaga ruticilla</i>	Group1	29
American Robin	<i>Turdus migratorius</i>	Group1	182
Bald Eagle	<i>Haliaeetus leucocephalus</i>		2
Bay-breasted Warbler	<i>Setophaga castanea</i>		17
Belted Kingfisher	<i>Megasceryle alcyon</i>		4
Black-and-white Warbler	<i>Mniotilta varia</i>	Group1	178
Black-backed Woodpecker	<i>Picoides arcticus</i>	Group2	23
Blackburnian Warbler	<i>Setophaga fusca</i>	Group1	227
Black-capped Chickadee	<i>Poecile atricapillus</i>	Group2	32
Black-throated Blue Warbler	<i>Setophaga caerulescens</i>		9

Black-throated Green Warbler	<i>Setophaga virens</i>	Group1	53
Blue Jay	<i>Cyanocitta cristata</i>	Group2	343
Blue-headed Vireo	<i>Vireo solitarius</i>	Group1	79
Boreal Chickadee	<i>Poecile hudsonicus</i>	Group2	34
Broad-winged Hawk	<i>Buteo platypterus</i>		3
Brown Creeper	<i>Certhia americana</i>	Group1	76
Canada Goose	<i>Branta canadensis</i>		39
Canada Jay	<i>Perisoreus canadensis</i>	Group2	92
Canada Warbler	<i>Cardellina canadensis</i>	Group1	117
Cape May Warbler	<i>Setophaga tigrina</i>	Group1	148
Cedar Waxwing	<i>Bombycilla cedrorum</i>	Group2	82
Chestnut-sided Warbler	<i>Setophaga pensylvanica</i>	Group1	215
Chipping Sparrow	<i>Spizella passerina</i>	Group1	153
Clay-colored Sparrow	<i>Spizella pallida</i>		2
Common Goldeneye	<i>Bucephala clangula</i>		2
Common Grackle	<i>Quiscalus quiscula</i>		16
Common Loon	<i>Gavia immer</i>		23
Common Merganser	<i>Mergus merganser</i>		2
Common Raven	<i>Corvus corax</i>		24
Common Yellowthroat	<i>Geothlypis trichas</i>	Group1	245
Connecticut Warbler	<i>Oporornis agilis</i>	Group1	29
Dark-eyed Junco	<i>Junco hyemalis</i>	Group1	86
Downy Woodpecker	<i>Dryobates pubescens</i>		9
Eastern Bluebird	<i>Sialia sialis</i>		7
Evening Grosbeak	<i>Coccothraustes vespertinus</i>		33
Golden-crowned Kinglet	<i>Regulus satrapa</i>	Group1	278
Great Crested Flycatcher	<i>Myiarchus crinitus</i>		13
Great Horned Owl	<i>Bubo virginianus</i>		1
Hairy Woodpecker	<i>Dryobates villosus</i>		32
Hermit Thrush	<i>Catharus guttatus</i>	Group1	217
Hooded Merganser	<i>Lophodytes cucullatus</i>		2
House Wren	<i>Troglodytes aedon</i>	Group1	21
Killdeer	<i>Charadrius vociferus</i>		3
LeConte's Sparrow	<i>Ammospiza leconteii</i>		5
Least Flycatcher	<i>Empidonax minimus</i>	Group1	63
Lincoln's Sparrow	<i>Melospiza lincolni</i>	Group1	82
Magnolia Warbler	<i>Setophaga magnolia</i>	Group1	575
Mallard	<i>Anas platyrhynchos</i>		10
Merlin	<i>Falco columbarius</i>		7
Mourning Dove	<i>Zenaidura macroura</i>		1
Mourning Warbler	<i>Geothlypis philadelphia</i>	Group1	191
Nashville Warbler	<i>Oreothlypis ruficapilla</i>	Group1	1556
Northern Flicker	<i>Colaptes auratus</i>	Group2	73
Northern Hawk Owl	<i>Surnia ulula</i>		1
Northern Parula	<i>Setophaga americana</i>	Group1	98
Northern Saw-whet Owl	<i>Aegolius acadicus</i>		1
Northern Waterthrush	<i>Parkesia noveboracensis</i>	Group1	66
Olive-sided Flycatcher	<i>Contopus cooperi</i>	Group1	64
Osprey	<i>Pandion haliaetus</i>		1
Ovenbird	<i>Seiurus aurocapilla</i>	Group1	551
Palm Warbler	<i>Setophaga palmarum</i>		20
Pileated Woodpecker	<i>Dryocopus pileatus</i>		23
Pine Siskin	<i>Spinus pinus</i>		68
Pine Warbler	<i>Setophaga pinus</i>		8
Purple Finch	<i>Haemorhous purpureus</i>	Group1	29
Red Crossbill	<i>Loxia curvirostra</i>		2
Red-breasted Nuthatch	<i>Sitta canadensis</i>	Group2	160
Red-eyed Vireo	<i>Vireo olivaceus</i>	Group1	209
Red-winged Blackbird	<i>Agelaius phoeniceus</i>		21
Rose-breasted Grosbeak	<i>Pheucticus ludovicianus</i>	Group1	41
Ruby-crowned Kinglet	<i>Regulus calendula</i>	Group1	355
Ruby-throated Hummingbird	<i>Archilochus colubris</i>		4
Ruffed Grouse	<i>Bonasa umbellus</i>		65
Sandhill Crane	<i>Antigone canadensis</i>		6
Scarlet Tanager	<i>Piranga olivacea</i>		2
Song Sparrow	<i>Melospiza melodia</i>	Group1	123
Sora	<i>Porzana carolina</i>		1
Spruce Grouse	<i>Falcapennis canadensis</i>		7
Swainson's Thrush	<i>Catharus ustulatus</i>	Group1	174
Swamp Sparrow	<i>Melospiza georgiana</i>	Group1	175
Tennessee Warbler	<i>Oreothlypis peregrina</i>	Group1	171
Tree Swallow	<i>Tachycineta bicolor</i>		4
Trumpeter Swan	<i>Cygnus buccinator</i>		1
Veery	<i>Catharus fuscescens</i>	Group1	63
White-throated Sparrow	<i>Zonotrichia albicollis</i>	Group1	1503
White-winged Crossbill	<i>Loxia leucoptera</i>		1
Wilson's Snipe	<i>Gallinago delicata</i>		40
Wilson's Warbler	<i>Cardellina pusilla</i>		9
Winter Wren	<i>Troglodytes hiemalis</i>	Group1	325
Yellow Warbler	<i>Setophaga petechia</i>		1

Yellow-bellied Flycatcher	<i>Empidonax flaviventris</i>	Group1	379
Yellow-bellied Sapsucker	<i>Sphyrapicus varius</i>		42
Yellow-rumped Warbler	<i>Setophaga coronata</i>	Group1	437

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