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Article

The Heterogeneity of Burn Severity Affects Bird Density in an Abandoned Mountain Landscape of the Atlantic-Mediterranean Transition

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Abstract: Fire regimes in mountain landscapes of southern Europe have been shifting from their baselines due to the accumulation of fuel fostered by long-standing rural abandonment and fire exclusion policies. Understanding the role of fire on biodiversity is paramount to implement adequate management to mitigate the impacts of altered fire regimes and land abandonment on biodiversity. Here, we explored to what extent the spatiotemporal variation in burn severity has affected bird abundance of a mountain abandoned landscape located in the Atlantic-Mediterranean transition (NW Iberia). We took advantage of: (1) satellite images of Sentinel 2 and Landsat missions to compute burn severity indicators from 2010 to 2020, and (2) standardized bird surveys carried out over 206 point-counts along the breeding season of 2021. Bird abundance models were built from burn severity metrics together with well-known fire regime attributes (% of burnt area and time since fire). Our results showed that the spatiotemporal variation of burn severity significantly correlated with the abundance of the 39% of the modeled species, supporting the role of *pyro*-diversity in driving bird populations in our region. The burnt area also explained abundance patterns for 28% of species. Time since fire only correlated with the abundance of 3 species. Our findings confirm the importance of incorporating burn severity indicators into the toolkit of decision makers to anticipate the response of birds to fire management.

Keywords: Burnt severity index; bird responses; generalized linear models; fire recurrency; time since last fire; Sentinel 2, Landsat satellite mission.

1. Introduction

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Fire is a major ecological and evolutionary driver of biodiversity in Mediterranean-type ecosystems [1]. The impact of fire and its management on biodiversity has been evaluated across different socio-ecological and biogeographic contexts over the last decade [2–5]. However, land managers are facing new challenges due to the effects of global change on fire regimes [2], an issue of paramount importance for endangered species and protected areas across the globe (see e.g. [6–8]). In rural landscapes of southern Europe, fire regimes have been gradually shifting from their baselines due to climate change and the accumulation of fuel caused by long-standing agricultural abandonment and fire exclusion policies [9,10]. Understanding the role of fire on biodiversity is paramount for decision makers to implement adequate management actions to halt biodiversity loss across protected areas [6,11]. The lack of a deep knowledge of the possible and complex responses of biodiversity to fire undermines the capacity of managers to decide when, where and how to implement fire management.

Over the last decade, ecologists have studied the effects of fire on biodiversity across different taxa (e.g., plants, reptiles and birds [12–14]), species traits [15,16], pre-fire conditions [17,18], and biogeographic contexts [4,19–21]. However, the variables used to explore the role of fire on biodiversity have been mostly focused on the area affected by fire, and the time since the fire took place (as *surrogate* of post-fire vegetation regeneration and patch-mosaic configurations [18,22,23]). Despite being a critical descriptor of wildfire intensity and damage, the effect of burn severity on biodiversity is less studied (see meta-analysis in [24]). To what extent the heterogeneity (i.e., the spatiotemporal variation) of burning affects biodiversity patterns in abandoned mountain landscapes remains largely unknown —a critical issue under anthropogenic fire regimes where species are not particularly well adapted to fire.

Herein, we assess the effect of wildfires on bird communities in a biogeographic transition zone between the Eurosiberian and Mediterranean region (the Natural Park 'Baixa Limia - Serra do Xurés', in NW Iberia). This area is a mountain range, strongly affected by rural abandonment and wildfires over the last decades [25]. Considering the large number of fires and the rapid post-fire vegetation recovery, we expect that fire has largely influenced bird abundance patterns in the region. We explore to what extent the spatiotemporal variation in burn severity has affected the abundance of birds in relation to other well-known fire regime attributes (such as percentage of burnt area, time since last fire and fire recurrence [see e.g. [23,26,27]). We took advantage of the satellite images of Sentinel 2 and Landsat missions (freely available from European Space Agency and NASA, respectively) to compute burn severity indicators over the last 11 years (from 2010 to 2020) and a standardized bird survey carried out along the breeding season of year 2021 to explore those relationships.

2. Materials and Methods

2.2. Study area and fire regime

This study was carried out in the Natural Park 'Baixa Limia-Serra do Xurés' (29,345 ha), a mountain area located in the south-west of the province of Ourense (Galicia, NW Spain, 42°–8°; **Fig. 1**), included in the Transboundary Biosphere Reserve 'Gerês-Xurés' [28]. The elevation ranges from 323 to 1,529 m a.s.l. with an average slope of 13°. The region is in the transition between the Mediterranean and Eurosiberian biogeographic zones in the proximity of the Atlantic coast. The climate regime is temperate oceanic sub-Mediterranean, with an average maximum temperature during the hottest month (June) of 22.8 °C and an average minimum temperature during the coldest month (January) of 0.29 °C. Average annual rainfall is 1,223 mm [29]. The study area has been subjected to

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land abandonment processes (i.e. vegetation encroachment and forest spread) since the second part of the last century due to rural exodus [25].

The fire regime in the study area is characterized by a very large number of small- and medium-sized fires [30]. Most of these fires are caused by human activity and classified as arson (87%). The land covers most affected by fire in the 'Gerês-Xurés' between 2000 and 2010 were sparsely vegetated areas and closed shrubland (49 and 15 % respectively), followed by pine plantations and oak woodlands (8.8 and 8.7 %) [25]. The speed of initial post-fire recovery is related to differences in fire-response traits of vegetation and to climatic conditions immediately following fire. The mid-term recovery indicator is mainly influenced by fire traits and post-fire climatic conditions. However, the long-term recovery is more influenced by burn severity than by vegetation type and structure or by post-fire climatic conditions [31].

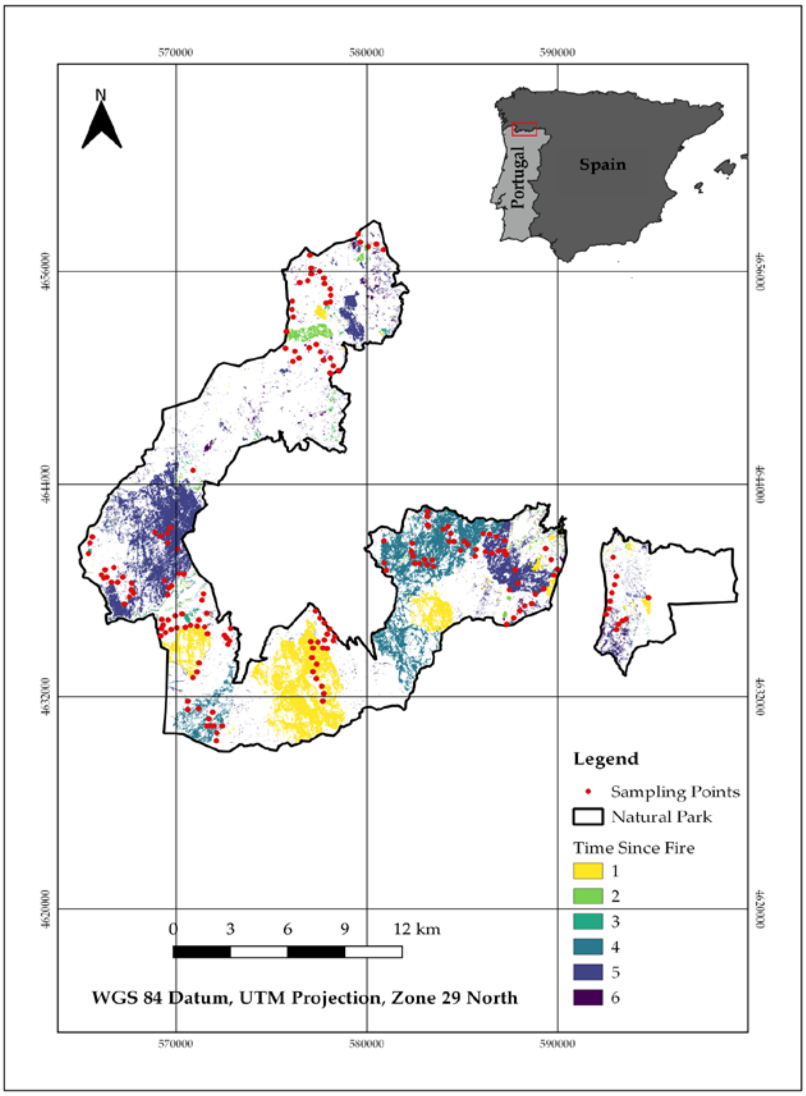

Figure 1. Location of the Natural Park 'Baixa Limia-Serra do Xurés' and the Time Since last Fire for the period 2010-2020. Red dots represent the point counts used to estimate bird density at species level.

2.2. Bird abundance

The bird community was surveyed by means of 5-min point counts (206 sampling units, Fig. 1) with unlimited distance. The censuses were undertaken during the breeding season (from mid-May to mid-July) of the year 2021. The censuses were carried out during the 4 h after sunrise (peak vocal activity) and under uniform weather conditions (days without marked rainfall or wind) to avoid possible detection biases caused by the time of survey, wind speed or cloud cover [32]. Each plot was considered to cover an area of about 3 ha, as most individuals were recorded within a radius of 100 m from the point count. This census method is designed to provide information on abundance of diurnal songbirds [32]. Raptors, crepuscular species and aerial feeders (mainly swallows and swifts) were not considered in the analysis since this census method is not suitable for those species.

2.3. Fire regime attributes

We created a set of fire-related variables to characterize the fire regime. Those fire-related variables were based on burnt severity indices derived from satellite imagery and the official database of fire scars perimeters of the Galician Government (Xunta de Galicia). For each year between 2010 and 2020, we used a pair of satellite images, one before (April - July) and one after (September - November) the fire season. To use the best available information, we selected different satellites along the study period: for years 2010 and 2011, we used Landsat 5 imagery, whereas for 2012 we used Landsat 7, since Landsat 5 imagery was not useful due to high cloudiness. Since its launch in 2013, we shifted to Landsat 8 data, and finally to Sentinel 2 data from 2015 onwards (**Table 1**). For each of these images we calculated the Normalized Burn Ratio (NBR), which is the normalized ratio between near-infrared (NIR) and short-wave infrared (SWIR) radiation **(Eq. 1).** NIR and SWIR bands of satellite sensors respond in opposite ways to burned vegetation, allowing to identify burned areas [33].

$$NBR = (NIR - SWIR) / (NIR + SWIR) \tag{1}$$

To obtain a quantitative measure of change for each year, we calculated the dNBR by subtracting the NBR of the post-fire season image from the NBR of the pre-fire season image **(Eq. 2).** Finally, dNBR values were used as an estimate for fire severity (see Fig. 2).

$$\boldsymbol{dNBR = NBR_{prefire} - NBR_{postfire}} \tag{2}$$

Table 1. Satellite sensors used for each year, spatial resolution, and the corresponding NIR and SWIR bands used to calculate the Normalized Burn Ratio (NBR).

Satellite	Resolution	Years	NIR band	SWIR band
Landsat-5 TM	30 m	2010 - 2011	B4	B7
Landsat-7 ETM+	30 m	2012	B4	B7
Landsat-8 OLI/TIRS	30 m	2013 - 2014	B5	B7
Sentinel-2	10 m	2015 - 2021	B8	B12

We computed the average, minimum, and maximum values within the buffer of 100 meter around each point count to characterize the burn severity along the 11-year period at each census plot. The range (i.e., the difference between the maximum and minimum

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value) was then calculated to measure the spatiotemporal variation of burn severity within the census plot (as *proxy* for pyro-diversity; *sensu* [34]). In addition, we estimated the percentage of burnt area in each census plot from the fire scars, available in vectorial format from official data of the Galicia Government (Xunta de Galicia) for the 2010-2020 period. Vectorial data was rasterized at the same spatial resolution as satellite images. To characterize other important attributes of fire regime over that period, we calculated fire recurrence (i.e., times each plot was affected by a fire event) and time since the last fire (from 1 to 11 years).

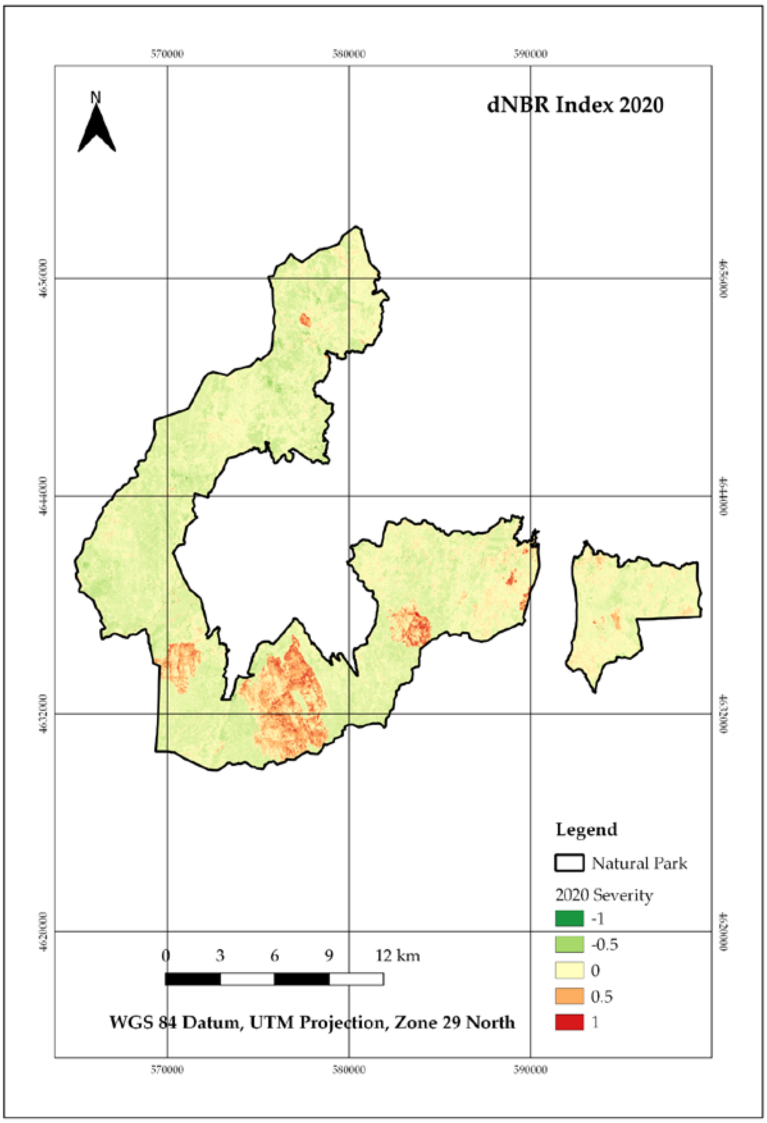

Figure 2. dNBR Index for the wildfires that took place in the Natural Park 'Baixa Limia-Serra do Xurés' in 2020 (see areas affected by those wildfires at census-plot scale from UAV images at Video S1).

2.4. Effects of fire regime on bird abundance

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To assess the effects of fire regime on the abundance of each target species (N = 28; Table 2), we fitted generalized linear models [35]. Since the response variable (i.e., bird density measured through a relative abundance index) represents counts of individuals, our models were fitted by using three types of approaches: GLMs with Poisson error distribution, GLMs with negative binomial error distribution, and Zero-inflated models. The best model for each target species was selected according to the explanatory power (measured by the pseudo-R^2) and the dispersion parameter (deviance/degrees of freedom), which should approximate to 1.0 to assume no overdispersion within the models. The analyses were performed with the 'MASS' R package. The variance inflation factor (VIF) was also calculated with the 'usdm' R package [36] to estimate how much a regression coefficient's variance is inflated by multicollinearity. The variables with VIF value greater than 3 were excluded from further analyses [37].

3. Results

The good-of-fitness of the bird abundance models (i.e., the proportion of the variation in the dependent variable that is predictable from the independent variables) was relatively low for most of our target species (pseudo-$R^2_{\text{-average}}$ of 0.10 ± 0.07, Table 2). Overall, GLMs with Negative Binomial or Poisson error distributions overperformed zero-inflated models (see Table 2). GLMs fitted using Poisson distribution yielded slightly better explanatory power. However, Poisson GLMs showed overdispersion (i.e., residual deviance divided by the degrees of freedom is larger than 1; see Table 2), so Negative binomial GLMs were finally selected as the best modeling approach for our target species.

3.1. Linear relationships between bird abundance and fire regime

According to the Negative Binomial GLMs, the abundance of 57% of the target species (16/28) were significatively affected by at least one attribute of the fire regime (hereafter: 'fire-sensitive' species) (Table 3). The abundance of 14% of species (5/28) was found to be affected by two of the three explanatory variables, being the Dartford warbler (*Sylvia undata*) the only one whose abundance was influenced by all three variables (Table 3). The spatiotemporal variation of burn severity was the most important factor affecting the abundance patterns for 39% of the modeled species (11/28), followed by the percentage of burnt area that significatively affected 28% of species (8/28) (Table 3). The models showed that the spatiotemporal variation of burn severity increases the abundance of most of the fire-sensitive species (9 of 11 species sensitive to fire severity, Table 3). The burnt area was found to increase the abundance of half of the species affected by the burnt area (4/8) (Table 3). The time since the last fire was found to positively affect the abundance of 3 species (Table 3).

3.2. Non-linear relationships between bird abundance and fire regime

For several birds, the relationship between their abundance and fire regime attributes was also nonlinear, even though the explanatory power of second-order polynomial GLMs was clearly lower than assuming linear relationships (Table 2). The abundance of 46% of the modeled species were non-linearly influenced by at least one attribute of the fire regime (Table 3). Only 2 species were affected by two of the three explanatory variables (Table 3). The non-linear relationship between spatiotemporal variation of burn severity and bird abundance was statistically significant for 35% of the modeled species, followed by the percentage of burnt area which significatively affected 17% of species (Table 3). The models showed that the spatiotemporal variation of burn severity increases the abundance of most of the fire-sensitive species (8 of 10 species sensitive to burn severity, Table 3). The polynomial relationship between bird abundance and burnt area was positive for 4 of the 5 species affected by the burnt area (Table 3).

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Table 2. Pseudo-R^2 and dispersion parameter (residual deviance divided by the degrees of freedom, 'Disp') for linear and nonlinear GLMs (I) fitted with Negative Binomial (NB) and Poisson error distribution, and zero-inflated models. Scientific species names in Table 3.

Common name	R^2	Disp	R^2	Disp	R^2	Disp	R^2	Disp	R^2	Disp	R^2	Disp
	NB		NB (I)		Poisson		Poisson (I)		Zero Inflated		Zero Inflated (I)	
Common Wood pigeon	0.089	0.973	0.091	0.999	0.085	1.130	0.088	1.148	0.004	0.925	0.005	0.936
Great spotted woodpecker	0.060	0.937	0.047	0.994	0.053	1.035	0.042	1.092	0.005	1.036	0.002	0.976
woodlark	0.113	0.930	0.122	0.966	0.105	1.174	0.114	1.194	0.010	1.092	0.013	0.916
Skylark	0.070	0.920	0.069	0.880	0.103	2.029	0.099	1.961	0.022	1.041	0.025	0.983
Eurasian wren	0.052	1.013	0.062	1.018	0.054	1.080	0.064	1.066	0.032	0.956	0.038	0.951
Dunnock	0.031	0.818	0.036	0.814	0.031	0.818	0.036	0.814	0.017	0.761	0.010	0.818
European robin	0.108	0.952	0.079	0.958	0.106	1.093	0.078	1.116	0.035	0.874	0.030	0.892
European stonechat	0.086	0.978	0.059	0.940	0.108	1.555	0.074	1.568	0.055	1.057	0.050	1.060
Common blackbird	0.043	0.914	0.031	0.924	0.043	0.914	0.031	0.924	0.028	0.853	0.023	0.850
Song thrush	0.125	0.855	0.110	0.932	0.121	0.908	0.107	0.986	0.017	0.841	0.010	0.895
Mistle thrush	0.061	0.630	0.077	0.625	0.080	4.843	0.144	4.657	0.023	1.167	0.016	1.351
Dartford warbler	0.131	1.005	0.092	0.999	0.171	1.626	0.125	1.690	0.117	1.052	0.090	1.060
Common whitethroat	0.030	1.034	0.019	1.053	0.025	1.652	0.016	1.684	0.003	0.951	0.002	0.986
Eurasian blackcap	0.138	0.999	0.078	0.937	0.143	1.142	0.078	1.143	0.032	0.981	0.021	0.929
Western Bonelli's warbler	0.161	0.689	0.148	0.737	0.164	1.732	0.164	1.830	0.034	0.832	0.047	0.934
Iberian chiffchaff	0.040	1.018	0.025	1.023	0.038	1.089	0.024	1.103	0.006	0.835	0.005	0.907
Common firecrest	0.054	0.742	0.041	0.711	0.068	2.125	0.067	2.210	0.019	0.884	0.021	0.904
Coal tit	0.184	1.022	0.174	1.342	0.204	1.376	0.198	1.851	0.075	0.919	0.052	1.152
Blue tit	0.103	1.626	0.035	1.045	0.105	2.616	0.032	1.616	0.002	1.830	0.001	1.062
Great tit	0.137	0.963	0.104	0.927	0.137	0.963	0.104	0.927	0.015	0.513	0.013	0.506
Red-backed shrike	0.025	0.884	0.025	0.915	0.026	1.200	0.020	1.241	0.004	0.715	0.006	0.719
Eurasian jay	0.226	0.773	0.182	0.773	0.193	1.052	0.151	1.098	0.029	0.870	0.032	0.895
Carrion crow	0.379	0.915	0.339	0.865	0.360	2.394	0.330	2.511	0.062	1.191	0.227	0.891
Common chaffinch	0.052	1.039	0.063	1.111	0.064	2.705	0.091	2.791	0.047	1.272	0.065	1.353
European serin	0.162	0.869	0.143	0.850	0.175	1.540	0.154	1.547	0.035	0.945	0.032	0.941
Greenfinch	0.067	0.919	0.050	0.925	0.061	1.456	0.044	1.475	0.003	0.932	0.002	0.880
Common Linnet	0.097	0.850	0.098	0.832	0.158	1.741	0.161	1.704	0.088	0.918	0.115	0.889
Rock bunting	0.107	0.999	0.107	0.966	0.108	1.172	0.105	1.151	0.021	0.947	0.024	0.968
$\bar{x}$	0.105	0.938	0.090	0.931	0.110	1.577	0.098	1.575	0.030	0.971	0.035	0.950

Table 3. Results of the Generalized Linear Models (**GLMs**) fitted using **Negative Binomial** (NB) distribution assuming linear and non-linear (I) relationships (quadratic effect; ^2) between bird density (relative abundance; birds/min) and: (1) the spatiotemporal variation of burn severity (**Sev_range**), (2) the percentage of burnt area

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within 100-m buffer sound each point count (**AB_mean**), and (3) the time since the last fire (**TSF**).

Common name	Scientific name	NB			NB (I)		
		Sev_range	AB_mean	TSF	Sev_range^2	AB_mean^2	TSF^2
Common Wood pigeon	*Columba palumbus*	**4.666 (2.197)***	-	-	3.944 (2.153)*	-	-
Great spotted woodpecker	*Dendrocopos major*	-	-	-	-	-	-
Woodlark	*Lullula arborea*	-5.066 (-2.117)*	-	-	-5.786 (-2.158)*	-	-
Skylark	*Alauda arvensis*	-	-	-	-	-	-
Eurasian wren	*Troglodytes troglodytes*	1.879 (2.522)*	-	-	1.852 (2.706)**	-	-
Dunnock	*Prunella modularis*	-	-	-	-	-	-
European robin	*Erithacus rubecula*	**3.126 (2.820)****	-	-	2.574 (2.542)*	-	-
European stonechat	*Saxicola rubicola*	-	10.119 (2.925)**	**0.090 (2.083)***	-	49.024 (2.429)*	-
Common blackbird	*Turdus merula*	-	-	-	-	-	-
Song thrush	*Turdus philomelos*	-	-	-	-	-	-
Mistle thrush	*Turdus viscivorus*	-	-	-	-	-	-
Dartford warbler	*Sylvia undata*	-2.587 (-2.796)**	9.037 (3.084)**	**0.071 (1.965)***	-2.491 (-2.569)*	36.070 (2.021)*	-
Common whitethroat	*Sylvia communis*	-	-	-	-	-	-
Eurasian blackcap	*Sylvia atricapilla*	**3.316 (2.868)****	-11.044 (-2.325)*	-	3.016 (2.724)**	-	-
Western Bonelli's warbler	*Phylloscopus bonelli*	-	-23.340 (-2.330)*	-	-	1.969e+02 (-1.972)*	-
Iberian chiffchaff	*Phylloscopus ibericus*	-	-	-	-	-	-
Common firecrest	*Regulus ignicapilla*	-	-	-	-	-	-
Coal tit	*Periparus ater*	-	-1.446 (-3.240)**	-	-	-1.245e+02 (-2.818)**	-
Blue tit	*Cyanistes caeruleus*	-	-	-	-	-	-
Great tit	*Parus major*	**5.833 (2.386)***	-	-	4.702 (2.274)*	-	-
Red-backed shrike	*Lanius collurio*	-	-	-	-	-	-
Eurasian jay	*Garrulus glandarius*	**7.091 (2.896)****	-	0.333 (2.358)*	5.800 (2.790)**	-	0.016 (2.422)*
Carrion crow	*Corvus corone*	6.978 (2.395)*	**-32.799 (-1.980)***	-	5.936(2.213)*	-	-
Common chaffinch	*Fringilla coelebs*	3.309 (2.830)**	-	-	2.935 (2.656)**	-	-
European serin	*Serinus serinus*	**3.549(1.991)***	-	-	-	-	-
Greenfinch	*Carduelis chloris*	-	-	-	-	-	-
Common Linnet	*Carduelis cannabina*	-	8.955 (2.201)*	-	-	55.122 (2.234)*	-
Rock bunting	*Emberiza cia*	-	9.909 (2.267)*	-	-	-	-

*p-value < 0.05; ** p-value < 0.01; ***p-value < 0.001.

4. Discussion

Our results showed that the spatiotemporal variation in burn severity (*proxy* for pyro-diversity) significantly correlated with bird abundance in our study area (see Table

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3). In relation to the effects of other well-known fire regime attributes (percentage of burnt area and time since last fire), the spatiotemporal variation in burn severity was found to be the most relevant factor explaining the abundance patterns of our target species (39% of the 28 modeled species, see Table 3). These results confirm the importance of accounting for burn severity when assessing the impact of wildfires on bird diversity (see e.g., [27,38]), which has clear implications for fire management (see meta-analysis in [24]). Overall, the spatiotemporal variation of burn severity was positively correlated with the abundance patterns for 9 species (Table 3), which can be explained by an increase in the habitat heterogeneity created by the wildfires [19,26,39]. In addition, burnt area was correlated with the abundance of 28% of species, with both positive and negative effects on bird abundance (see Table 3). Overall, open habitat dwelling and early successional species (see e.g., the Dartford warbler in Table 3) were found to be positively affected by the amount of area affected by fire, but negatively by burning heterogeneity. These results are in line with previous studies that suggest that early successional species have peaks of abundance around 4-8 years after fire in Mediterranean ecosystems [40,41]. Fire creates habitat conditions to allow species such as Dartford warbler to occupy areas previously lost due to vegetation encroachment or afforestation processes driven by the long-standing rural abandonment and fire suppression (see e.g., [42]). In fact, recent studies already suggested fire as a tool to enhance bird conservation in this area, since fire could offset the negative effects of land abandonment on ecotone and open-habitat species [43,44]. On the contrary, forest-dwelling species (see e.g., Eurasian blackcap or Western Bonelli's warbler in Table 3) were negatively affected by fire, being favored by an increasing heterogeneity in burn severity (i.e., unburnt patches; see [45]).

Despite the number of studies supporting 'time since fire' as a good predictor of bird abundance [22,46,47]), this variable only correlated significantly with the abundance of 3 species in our study area (Table 3). Time since fire is often used to delineate successional vegetation states, being a good proxy for post-fire recovery, fuel age or vegetation structure [48,49]. However, our study region is characterized by fast recovery rates [31] due to its bioclimatic position, which could justify the less important role of time since fire for explaining bird densities in relation to burnt area and severity indicators (see Table 3). Nevertheless, future studies in the region should include longer time series for burn severity and fire occurrence in their analyses. Short- and mid-term bird populations' response to fire can significatively differ from longer-term post-fire conditions [50]—an critical issue that is also context-specific.

The relationship between bird abundance and fire regime attributes is not always linear. Some studies have found more complex relationships between fire and bird abundance [18,23]. It is therefore important to consider non-linear effects of fire severity and time since fire to fully understand post-wildfire responses for a majority of bird species (see e.g., [18]). In our case, we found a quadratic effect of at least one of the three fire regime attributes on bird abundance for 46% of the modeled species (see Table 3). However, the explanatory power of these models was considerably lower than those fitted assuming linear relationships (see Table 2), which suggests that adding non-linear relationships in our model would not provide much more information, leading to collinearity problems. In this sense, including other well-known bird habitat descriptions (e.g., coverage of trees, distance to farmlands or slope, among others; [51–53]) in abundance models would strongly enhance their predictive capacity (as shown in [54,55] for colonization/extinction patterns). Our results showed the usefulness of burn severity indicators to explain bird abundance patterns, which clearly complements the information provided by well-known fire regime descriptors such as the amount of burnt area, time since fire or fire recurrence. The complex relationships found between each fire regime descriptor and the density of each species highlights the tailor-made management that is needed at the species level, and the role that fire could play in such management.

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5. Conclusions

The heterogeneity in burn severity (measured through the spatiotemporal variation of dNBR at plot level for the last 11 years) significatively correlated with bird density, being a factor more relevant than burnt area and time since fire to explain bird abundance in our study area. More than half of the 28 modeled species (58%) was found to be sensitive to fire, whose densities were correlated with at least one attribute of the fire regime. These results confirm the role that fire can play in the conservation of these species in mountain regions largely affected by rural abandonment. However, the biogeographic position of our study area, in the transition between the Eurosiberian and Mediterranean regions, calls for caution when extrapolating our results to other contexts, since the response of birds to fire relies on the post-fire vegetation recovery and pre-fire conditions, properties that differ from one ecosystem to another. Despite the context-specific response of birds, our findings confirm the importance of incorporating remotely sensed indicators of burn severity into the toolkit of decision makers to accurately anticipate the response of birds to fire management actions such as prescribed fire.

Supplementary Materials: Video S1: Areas affected by those wildfires at census-plot scale at the Natural Park taken by Dr. J Gonçalves and A.R. from an Unmanaged Aerial Vehicle; https://youtu.be/NnR_5l6pxbM.

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