Brief Report

Lack of Small-Scale Changes in Breeding Birds after a Fire: Does the Resilience of Cork Oaks Favor Rapid Recolonization in Suburban Wood Patches?

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Simple Summary: When we compared breeding bird communities in cork oak patches located before and after a fire event, we did not observe any significant changes in the density of territorial pairs or in the diversity metrics. This counterintuitive response may be due to the characteristics of cork oak (*Quercus suber*), a sclerophilous tree that is very resilient to fires and able to recover foliage in the following spring season, thus allowing rapid bird recolonization.

Abstract: Forest fires are disturbance events that can impact biological assemblages at multiple scales. In this study, the structures of breeding bird communities in cork oak patches located in an agro-mosaic suburban landscape of central Italy (Rome) were compared at the local scale with a fine-grained mapping method before (2018) and after (2023) a fire event occurred in July 2022. The analyses did not reveal any significant changes in the density of territorial pairs or in the diversity metrics, both univariate (Shannon–Wiener index, evenness, Margalef normalized richness) and bivariate (Whittaker and k-dominance plots, abundance/biomass curves) of diversity. Even when the guilds of strictly forest-related species were compared, no differences emerged before and after the fire. This counterintuitive phenomenon may be due to the characteristics of the dominant tree, the cork oak (*Quercus suber*), a sclerophilous tree that is very resilient to fires and able to recover foliage in the following spring season, thus allowing rapid bird recolonization. However, other small-scale phenomena (e.g., the 'crowding effect' and local dispersal of territorial pairs from remnant wood patches not affected by fire) may explain this lack of change in breeding bird density and diversity. Further studies should be carried out at larger spatial and temporal scales and at different levels of fire frequency and intensity to confirm these responses at the guild/community level in suburban cork oak wood patches.

Keywords: agroforestry mosaic; Mediterranean oak forests; guilds; diversity/dominance diagrams; abundance/biomass comparisons

1. Introduction

Forest ecosystems can be impacted by anthropogenic fires [1]. Worldwide, several studies have demonstrated how fires impact bird communities in extremely different ways: bird assemblages occurring in burned areas undergo structural alterations depending on the fire regime (extent, duration, frequency, and intensity), as well as other environmental factors, especially vegetation cover and its resilience following the passage of fire [2–8]. Such disruption in the structure of bird communities can lead to an increase in ecotonal and generalist species, which are linked to edge and disturbed habitats, to the detriment of specialized species, which are linked to stable and mature environments, such as forest

habitats [9–12]. Fires disrupt both foraging and nesting sites of many forest-specialized species, changing the patterns of niches and resources [9]. However, the effects and responses of birds are strongly scale-dependent and species-/context-specific [13–16].

Compared to temperate broad-leaved and coniferous woods (e.g., [17–22]), forest vegetation in the Mediterranean basin is different and often highly resistant and resilient to the effects of fire, as is the case for Mediterranean maquis forests and, more specifically, for cork oak (*Quercus suber*) forests. The bark of cork oak trees has a high insulating capacity and can reach a thickness of approximately 30 cm, thus protecting the cambium from the heat of fire [23,24]; see also [25] for central Italy. Therefore, cork oak forests recover quickly following a fire; however, this recovery depends on the disturbance regime [26].

In 'Campagna Romana', an agroforestry countryside area surrounding the city of Rome (central Italy; [27]), many cork oak patches occur. These remnants form interesting fragmented 'archipelagoes' of habitat islands (sensu [28]) of high ecological interest and conservation concern [29,30]. These landscape mosaics are often affected by anthropogenic disturbances linked to the suburban context and, among these disturbances, fires [31]. Since these are areas bordering urbanized sites, such fires (mainly small-size and accidental events linked to agricultural practices and influenced by socioeconomic conditions mainly linked to antisocial behaviors or linked to crop land management [32]) are quickly controlled, and the affected surfaces are always limited; however, at the local scale, they can still heavily alter forest cover and structure and related biological diversity [22,33]. Therefore, specific wildfire protection plans have been defined (e.g., for Rome, [34]).

In this study, which was carried out in a suburban agroforestry mosaic located in central Italy, we report data on the structure of breeding bird communities present before (2018) and after (2023) a fire event that occurred in the summer of 2022. The fire-affected patches were dominated by cork oaks that, beginning in the autumn following the event, presented recovery of leaf cover, despite the passage of the fire being particularly destructive at the local scale. Therefore, we hypothesized that species typically associated with the cork oak forest structure (forest-related guild) were not adversely affected by the fire.

2. Materials and Methods

2.1. Study Area

The study area, with a total extent of 38 hectares, consists of a residual corridor of the Roman countryside ('Campagna Romana'; southwestern Rome suburban area), located between two nature reserves: "Valle dei Casali" and "Tenuta dei Massimi" (Latium Regional law n. 29/1997; size area: 38 hectares; 41°51′52.6″ N 12°24′53.6″ E; [35]; Figure 1). This is an agroforest mosaic with a landscape matrix composed of croplands and uncultivated fields dominated by *Vulpio-Dasypiretum* and *Diplotaxio tenuifolii-Agropyretum repentis* [36–38], including three forest patches (10.05% of the total area) dominated (>76% of tree frequency; *n*= 349; Supplementary Materials Table S1) by cork oaks (*Quercus suber*; [39]).

At the edges of the forest patches, there are characteristic ecotonal species belonging to the Mediterranean scrub vegetation (mock privet *Phillyrea* sp., Mediterranean buckthorn *Rhamnus alaternus*, laurel *Laurus nobilis*, and olive tree *Olea europaea*; Supplementary Materials Table S1). Furthermore, there are small ecotonal patches of hygrophilous vegetation with exotic species (eucalyptus *Eucalyptus* sp., stone pine *Pinus pinea*, Chinese privet *Ligustrum lucidum*) surrounding ponds and abandoned rural houses (details in [39]).

In 17 July 2022, a fire of great intensity (canopy crown fire [40]; duration: approximately five hours) affected 57.32% of the cork oak-dominated patches (the only wood patches occurring at the study site), resulting in a marked reduction in tree foliage cover [41]). The fire, probably started by bales of hay placed in the recently mown fields, left the cork oak tree structure intact, burning only the foliage and the 2nd- and 3rd-level branches. The fire had a heavy impact on the undergrowth (Mediterranean scrub, *Rubus* sp.), which disappeared. Since autumn 2022, and mainly in the following season (2023), signs of vegetation recovery have been observed both in the undergrowth and in the arboreal cork oaks (>7.5 cm).

Figure 1. Map of the study area and its location in Latium (star, on the right), central Italy, showing the suburban landscape mosaic. Cork oak (*Quercus suber*) patches are dark green. The polygons in red indicate the burnt areas where the fire event occurred (July 2022).

2.2. Bird Surveys

A multivisit territory mapping method was used to study the breeding bird community, which involves a spatial recording of direct observations, vocalizations, song contacts and alarm calls of the territorial bird species locally present to define the number of breeding pairs [42,43].

Twenty-two surveys were carried out from 9 April to 30 May 2023 (research effort: appr. 40 h) in the first hours of the morning (08.00–11.00 a.m.). Every survey (n = 22) was carried out randomly inside the period along different paths. In any survey (almost 2 h/visit), we walked the whole area via different routes. The data obtained for the entire study area, without a subdivision into burnt and unburnt subareas, were compared with those published in the study carried out before the fire event (spring 2018; [39]), via the same paths. The data were separated into three sampling subperiods (II half of April, I half of May and II half of May) to test for possible phenological differences.

Each survey was carried out under clear or variable weather conditions in the absence of rain or strong winds, all of which could affect sampling [42,43]. Sampling was conducted by two of the authors (S.C., C.B.). Species-specific territories were obtained following the clustering procedure described in [42]. We considered a "territory" as a range area where a territorial species pair was considered to be breeding [42,43]. Species with crepuscular or nocturnal activity (e.g., Strigiformes and Caprimulgiformes) and individuals flying very high (> 25 m) were not considered. Compared with 2019, in 2023 we carried out a comparable research effort (in terms of the number of visits and time effort, i.e., approximately 40 h); one of the authors (C.B.) carried out the surveys both in 2019 and 2023 to avoid observer effects.

The data were analyzed at the community, guild, and species levels. At the species level, in our analyses, we considered only the territorial breeding species (therefore excluding nonterritorial/gregarious breeders, e.g., *Passer* sp., *Sturnus vulgaris*, and non-breeders as species in migratory transit). In this last case, we selected two ecological guilds: (i) forest species and (ii) open/edge species, including species linked to (i) strictly forest habitat types (w, in Table 1) and (ii) species linked to open habitats (croplands) and ecotones (wood/maquis/cropland fringe; e/o in Table 1).

For all species, we obtained (i) the ecological density (as D = breeding pairs/10 ha), i.e., the density calculated considering only the suitable habitat cover (e.g., cork oak patch size area for the forest birds and crop land/ecotone size areas for open/edge species), (ii) the consuming biomass (Cb; in g/10 ha), calculated as: $Cb = Scb^{0.7}$ [44], where Scb, or the standing crop biomass, is the total body mass of all censused individuals (in g/10 ha). This value is directly proportional to the energy removed by individuals from the environment [44]. We used more consuming biomass than standing crop biomass because the former better explained the specific variations in metabolic rhythm, which were related mainly to individual size [44].

Table 1. The bird community in the study area (only territorially breeding species; non-territorial species or solely migratory non-breeding species were not included), before (2023) and after (2018) the 2022 fire event. Species and ecological guilds (w: forest species; e/o: open/edge species), densities (D, in breeding pairs/10 ha), and relative frequencies (Fr$_D$, as D/N, where D is species density and N is the total species density), and consuming biomass (Cb, in grams, and relative frequencies, Fr$_{Cb}$) for each species are reported. Below: the values of the univariate diversity metrics at the community level: normalized richness (Margalef index; Dm), Shannon–Wiener diversity (H′) and evenness (J). Phenological status: s: sedentary; m: long-distance migrant.

Species	Guild	2018 (Before Fire)				2023 (After Fire)			
		D	Fr$_D$	Cb	Fr$_{Cb}$	D	Fr$_D$	Cb	Fr$_{Cb}$
Short-toed Treecreeper *Certhia brachydactyla* (s)	w	0.263	0.016	6.486	0.013	0.789	0.049	13.995	0.027
Zitting Cisticola *Cisticola juncidis* (s)	e/o	1.053	0.064	15.344	0.031	0.263	0.016	5.814	0.011
Eurasian Wren *Troglodytes troglodytes* (s)	w	0.263	0.016	6.509	0.013	0.789	0.049	14.045	0.027
Long-tailed Tit *Aegithalos caudatus* (s)	w	0.263	0.016	5.634	0.011	0.263	0.016	5.634	0.011
Melodious Warbler *Hippolais polyglotta* (m)	e/o	0.526	0.032	13.226	0.027	0.526	0.033	13.226	0.025
European Turtle Dove *Streptopelia turtur* (m)	w	0.263	0.016	48.310	0.097	0.526	0.033	78.480	0.150
European Green Woodpecker *Picus viridis* (s)	w	0.263	0.016	61.256	0.123	0.263	0.016	61.256	0.117
Great Tit *Parus major* (s)	w	1.579	0.096	40.238	0.081	0.526	0.033	18.649	0.036
Common Blackbird *Turdus merula* (s)	w	1.579	0.096	107.797	0.216	1.842	0.115	120.079	0.229
Eurasian Blue tit *Cyanistes caeruleus* (s)	w	1.316	0.080	23.465	0.047	2.368	0.148	35.408	0.068
Eurasian Blackcap *Sylvia atricapilla* (s)	w	3.947	0.240	67.434	0.135	2.895	0.180	54.274	0.104
Common Nightingale *Luscinia megarhynchos* (m)	e/o	2.500	0.152	59.089	0.119	2.895	0.180	65.475	0.125
Sardinian Warbler *Curruca melanocephala* (s)	e/o	2.632	0.160	43.596	0.087	2.105	0.131	37.291	0.071
Dm		2.902			2.902		2.919		2.919
H′		2.191			2.191		2.250		2.250
J		0.854			0.854		0.877		0.877

To assign species to different ecological guilds, we refer to [45–49]. To calculate the biomass values, the mean body mass values were obtained from [45–49]. When available, we used biomass data available for an area immediately surrounding (4 km away; Villa Pamphili urban park; [50]), and (iii) the relative frequencies were calculated both for density (fr$_D$, as D/N, where N is the total species density) and for biomass (fr$_{Cb}$, expressed as the ratio of specific Cb/total Cb).

2.3. Data Preparation and Statistical Methods

To compare the taxonomic diversity of samples of different sizes and to verify the representativeness of our data, individual rarefaction curves were generated. These curves were also obtained by dividing the data into three subsampling periods (halves II of April, I and II of May) to verify possible phenological differences inside the study period at the level of species richness and diversity.

Univariate diversity metrics were then calculated: (i) Shannon–Wiener diversity index (H′; [51]), as H′ = −Σ fr$_D$ [lnfr$_D$]; (ii) evenness index (J; [52]), as J = H′/H′max, where H′max = lnS (with S = number of detected species); and (iii) normalized Margalef richness (Dm; [53]), as Dm = (S−1)/lnN.

With respect to bivariate diversity metrics, the following analyses were performed: (i) k-dominance plots [54,55], which allowed us to understand the maturity and stress level of the bird community (and of the forest species guild) on the basis of the uniformity of the frequency distributions of the different species; and (ii) ABC curves (abundance–biomass comparison [56,57]; an application for birds [58,59]), which make it possible to obtain information on the level of stress at the community level, rank the cumulative frequencies both for density (i.e., the normalized abundance) and biomass, and obtain the related curves. When abundance and biomass curves are compared, we may obtain information on the level of relative dominance of large- vs. small-body-mass species with structural and ecological implications (e.g., on the role of anthropogenic disturbances affecting the assemblages). ABC curves are based on a general assumption that, in human-disturbed habitats, small-sized species (i.e., those with low body masses and low trophic levels) tend

to increase in abundance; consequently, the abundance curves approach an asymptote before the biomass curves do. Instead, in mature and undisturbed habitats, an opposite pattern may be observed, with the biomass curves accumulating before the abundance curves, indicating that a greater number of large-body-mass species of high trophic level occur in more complex and diverse assemblages [56,57]: under these conditions, the frequency of abundance is more evenly distributed than the frequency of biomass. Early cumulative abundance curves may indicate that the resources are used by a few dominant (i.e., more abundant) species with a broad spatial niche (i.e., generalists), whereas early cumulative biomass curves may indicate that species with a relatively high biomass largely occur in the assemblage (review in [60]).

To compare the relative frequencies of the ecological densities of each species between the two years, a χ^2 test was carried out. To test for differences between the pairs of frequencies obtained in the two years before and after the fire, we used analysis of covariance (ANCOVA) [61,62]. To perform the statistical analyses, we used PAST software version 1.34 [63].

3. Results

In the 2023 spring period, 752 total records belonging to 13 different territorial breeding birds were obtained, both for forest and open/edge species. In 2018 (before the fire), 565 records were obtained: 62.5 territories were recorded (density: 16.45 pairs/10 ha), 37 of which were forest species (density: 9.74) and 25.5 of which were open/edge species (density: 6.71). In 2023 (after the fire), 61 territories were recorded (density: 16.05), of which 39 belonged to forest species (density: 10.26) and 22 to open/edge species (density: 5.79). The data at the species level reported in Table 1 were compared only for the territorial breeding species.

The individual rarefaction curves show a cumulation highlighting how the sample of collected data is representative both in the spring period and in the three sampling subperiods (II half of April, I half of May and II half of May; Figure 2).

The univariate diversity metrics (Margalef normalized richness, Shannon–Wiener and evenness indices), which were calculated for the 2023 data (after fire), revealed medium-high species diversity and a uniform abundance among species, comparable with those calculated in 2018 (before fire, Table 1). The same results were obtained by dividing the data at the guild level (forest and open/edge species; Supplementary Materials Tables S2 and S3).

Differences among species-specific frequencies between 2018 (before) and 2023 (after fire), both at the community and guild levels, were not significant (Table 2 and Supplementary Material Table S4).

The Whittaker and k-dominance plots, which compare the frequencies obtained both from the ecological densities and biomasses at the community level and at the forest guild level, show a comparable evenness before (2018) and after (2023) the fire. The analysis of covariance did not reveal a significant difference between the patterns in frequency both for abundance (Whittaker plots—bird community level: F = 7.757, p = 0.994; forest guild level: F = 0.0001, p = 0.997; k-dominance plots—bird community level: F = 0.245, p = 0.625; forest guild level: F = 0.992, p = 0.335; ANCOVA test; Figure 3) and biomass (Whittaker plots—bird community level: F = 0.001, p = 0.992; forest guild level: F = 0.0001, p = 0.981; k-dominance plots—bird community level: F = 0.084, p = 0.775; forest guild level: F = 0.007, p = 0.935; ANCOVA test; Figure 4).

The ABC curves also show a large overlap between the two periods before (2018) and after (2023), highlighting an absence of significant differences between the two patterns (ANCOVA test, 2018: F = 0.461, p = 0.504; 2023: F = 0.015, p = 0.902. Figure 5).
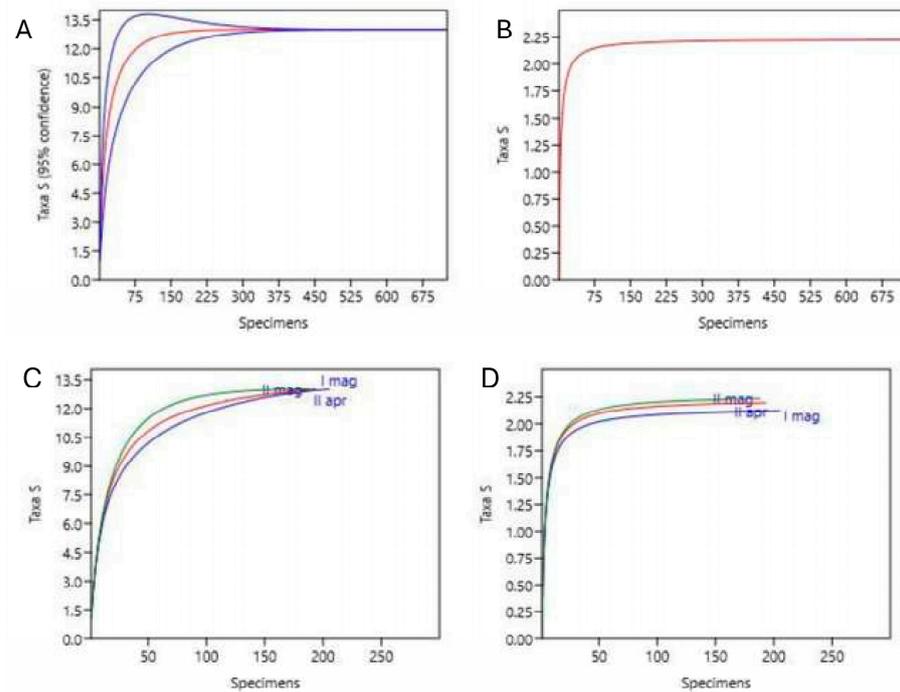
Figure 2. Individual rarefaction curves for specific richness (left) and the Shannon diversity index H′ (right) for the entire sampling period (top) and for the three subperiods represented separately (bottom). For richness, the 95% confidence intervals are reported (in blue). (**A**,**B**) respectively, specific richness and Shannon index, entire period (in red line; in blue the confidence intervals); (**C**,**D**) respectively, specific richness and Shannon index, three subperiods [(**C**): I half of May: green; II May: red; II April: blue; (**D**): II May: green; II April: red; I May: blue].
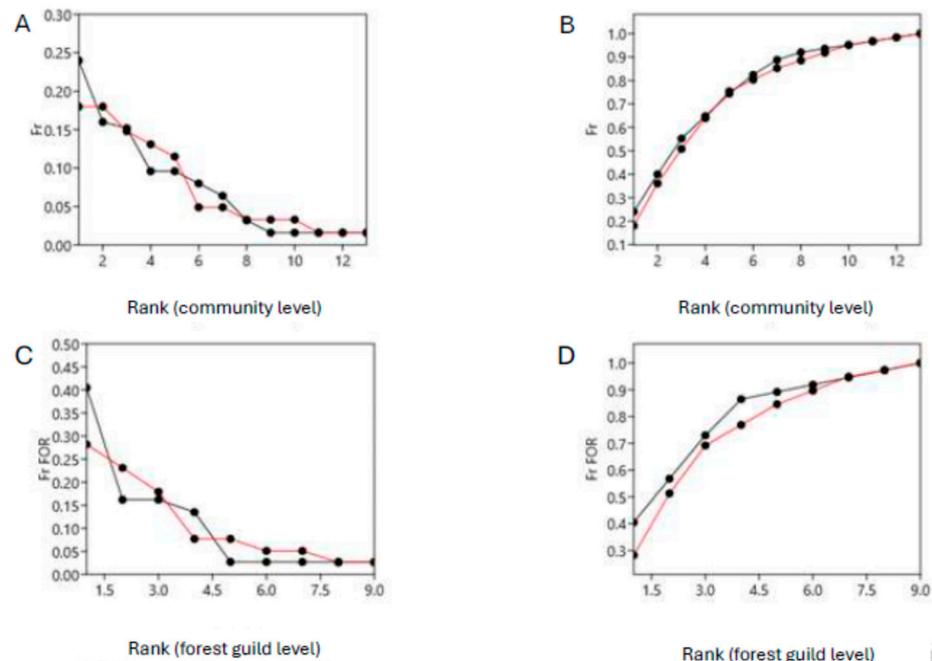
Figure 3. Whittaker (left) and k-dominance plots (right) made by comparing frequencies derived from the ecological densities of the entire bird community (above) and forest guild (below) before (2018, lines in black) and after (2023, lines in red) the fire event (July 2022). (**A**,**B**): respectively, Whittaker plots and k-dominance plots for the entire bird community; (**C**,**D**): respectively, Whittaker plots and k-dominance plots for the forest guild.

Table 2. Statistical comparisons between species frequencies (on density; Fr_D; as D/N, where D is species density and N is the total species density) calculated for the 2018 (before) and 2023 (after the 2022 fire event) data (with the χ^2 test and *p*-value). Only data on territorial breeding species have been reported (non-territorial species or migratory passes have not been included).

|	Fr_D 2018	Fr_D 2023	χ^2	*p*
Short-toed Treecreeper *Certhia brachydactyla*	0.016	0.049	0.284	0.594
Ztting Cisticola *Cisticola juncidis*	0.064	0.016	0.784	0.375
Eurasian Wren *Troglodytes troglodytes*	0.016	0.049	0.284	0.594
Long-tailed Tit *Aegithalos caudatus*	0.016	0.016	<0.001	1
Melodious Warbler *Hippolais polyglotta*	0.032	0.033	<0.001	1
Eurasian Turtle Dove *Streptopelia turtur*	0.016	0.033	0.0004	0.983
European Green Woodpecker *Picus viridis*	0.016	0.016	<0.001	1
Great Tit *Parus major*	0.096	0.033	1	0.288
Common Blackbird *Turdus merula*	0.096	0.115	0.002	0.963
Eurasian Blue Tit *Cyanistes caeruleus*	0.080	0.148	0.809	0.368
Eurasian Blackcap *Sylvia atricapilla*	0.240	0.180	0.351	0.553
Common Nightingale *Luscinia megarhynchos*	0.152	0.180	0.0328	0.856
Sardinian Warbler *Curruca melanocephala*	0.160	0.131	0.039	0.842
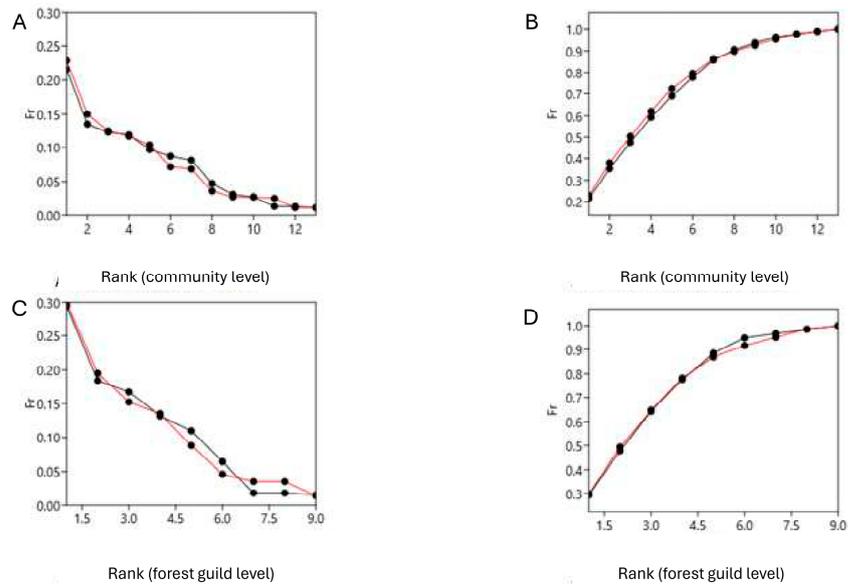
Figure 4. Whittaker (left) and k-dominance plots (right) made by comparing frequencies derived from the biomass of the entire bird community (above) and only for the forest guilds (below) before (2018, lines in black) and after (2023, lines in red) the fire event (July 2022). (**A**,**B**): respectively, Whittaker plots and k-dominance plots for the entire bird community; (**C**,**D**): respectively, Whittaker plots and k-dominance plots for the forest guild.
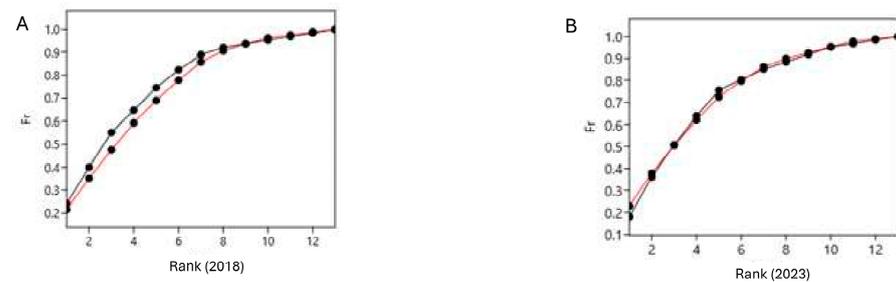
Figure 5. Abundance/biomass comparisons (ABC) curves for the 2018 (**A**) and 2023 data (**B**). Black lines: abundance curves; red lines: biomass curves.

4. Discussion

At the local scale, the structure and composition of territorial species of breeding bird communities did not significantly change between the two periods before (2018) and after (2023) the 2022 fire event. Although an increase in open/edge birds was expected to be detrimental to forest species, the passage of the fire had no impact either at the community level or at the level of two selected guilds (forest and open/edge species). Our results are counterintuitive and different from those of many studies focusing on changes in abundance, richness and diversity where abrupt reductions in these metrics have been observed after fires (e.g., [64,65]).

Cork oaks are highly fire-resilient trees with recovery capacity already in the season following the fire, being the only European tree with stem and crown resprouting capability (through epicormic buds) after intense crown fires [66–69]. In our case study, the post-fire crown regeneration of the tree canopy, which began in the autumn following the fire, and the vegetative recovery of cork oak trees in the following spring (2023) have probably restored the habitat suitability for forest birds (see [70] for Algerian cork oak woods). Since this patch system occurs in a fragmented suburban agroforestry landscape, it is likely that other factors may have contributed to this lack of expected changes in our bird assemblages. For example, it is conceivable that, following the fire, an increase in density occurred in the residual areas not affected by the fire. This phenomenon, known as the 'crowding effect' [71], occurs in fragmented landscapes when specific disturbance-induced changes lead to an abrupt reduction in the size of suitable habitat for sensitive species (e.g., [72]). In fragmented agroforestry landscapes, it is also probable that in the presence of patches not affected by disturbances, rapid recolonization processes toward the burned areas occurred, thus re-establishing, in the short term, the structure and composition of the breeding bird communities (particularly in forest-related species).

According to Mendelsohn et al. [73], our data suggest that the observed responses of birds to fires may be attributed to (i) the availability of nearby unburned habitat patches (showing a role of refugia), (ii) the local suitability of post-fire resilient vegetation, and (iii) the species-specific dispersal capacity.

However, this is only an exploratory study, highlighting the post-fire resilience of breeding birds in suburban cork oak wood patches at a local scale and in a short time range. In this context, our considerations may not be valid in spatial and historical circumstances where fires act locally, with high frequency and intensity (e.g., as 'pulse' events; [74]). For example, Touhami et al. [75] highlighted how cork oak forests can severely degrade when fires and related disturbances (e.g., grazing) occur at a rate that does not allow for the recovery of forest systems (i.e., as 'press' disturbances [74]; Zeghdani et al. [70] showed long times to restore the pre-fire densities also for forest species).

Moreover, it is important to highlight some limitations of this study. First, since some species recorded are Eurasian or Eurasian/African (trans-Saharan) migrants, these species could be affected by large-scale (i.e., not local) events (climate conditions, availability of food sources, other disturbances) along their migratory routes [76]. Therefore, the observed lack of changes may not be directly attributable to local fire events.

Second, during the time gap between the pre-fire (2018) and post-fire (2023) periods, several changes unrelated to the fire could have occurred, affecting the bird communities. Without continuous monitoring throughout this period or at least more frequent surveys and given the lack of 'control' areas (i.e., comparable sites without fires), it is difficult to assign the observed changes (or lack of changes) directly to the fire event. However, although 'Campagna Romana' has undergone major transformations in recent decades [77,78], the local pattern of landscape mosaics has not changed in the last 2–5 years (pers. obs.). Our study area has been characterized by agro-mosaic stability over time (i.e., with few, recurrent and comparable disturbances occurring any year; the only other anthropogenic disturbance occurring locally is crop mowing in June). Therefore, we are confident that from 2018 to June 2022 (before fire), bird communities were very similar in structure, and we may consider our design as a before/after comparison.

Third, our local data at the species level could be affected by population trends at larger scales (i.e., national or regional ones). All the species here considered have shown a stable or increasing trend in the first decades of the 2000s (except the Eurasian Turtle Dove and Zitting Cisticola, which show stable/decreasing trends) and, particularly, the guild of forest species considered both individually and as a whole (see the Woodland Bird Index; for Europe: +7% from 2010 to 2022 [79]; for Italy: [80,81]; for Latium [82]; see also [83] for Northern Italy). Therefore, our conclusions may not be extendable to other species showing declining trends. However, since data in population size at national level population sizes were comparable in both study years, we think that our post-fire results indicate the impacts (or lack of impact) of fire and do not depend on other factors on a larger scale.

Finally, we carried out a study at the community and guild levels, assuming that all species associated with the forest complex were uniformly impacted by the fire. This represents an oversimplification, and further analyses are necessary at the single-species level. For example, some passerines (e.g., *Curruca melanocephala* and *Hippolais polyglotta*) show interesting responses in the first year after fires (e.g., [13,14]).

Therefore, further studies should confirm the fire responses at the species and community levels in suburban forest mosaics dominated by epicormic resprouting trees (such as, in our case, cork oaks), which are largely diffused worldwide (e.g., [84]), and at larger spatial and temporal scales for different disturbance regimes and use guilds obtained with different criteria (e.g., foraging- and phenological-based).

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