Bird community responses to savanna fires: should managers be concerned?

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Despite fire being regarded as an important process for driving and maintaining ecological diversity, its influence on animal communities is poorly understood. This study investigates medium-term consequences of a savanna fire on bird communities in the Kruger National Park, by comparing mild/cool and severe/hot sections of a burn, and control (unburned) sites. The number of species recorded did not differ significantly between treatments, although mild sites, when matched with control sites, had a marginally higher species count. Mild fires may increase the diversity of habitats available to birds. Differences in bird community composition by dietary group were minor, with some best explained by pre-fire habitat differences. However, some differences were linked to habitat change as a result of fire. Granivores and ground-feeding species that prefer cover were less common on severe burns, where grass cover was most reduced. However, no species were entirely absent. In savannas, fire has a short return time and post-fire habitats recover rapidly compared to other fire-prone ecosystems; not even severe fires appear to disturb bird communities significantly. They are likely to be robust to all but the most extreme fire policies, which suppress fire completely or alter habitat structure. A hands-off fire policy is unlikely to affect bird communities negatively.

Key words: birds, communities, ecology, fire, savannas.

INTRODUCTION

Fire is one of the most ubiquitous agents of disturbance in natural systems and influences the evolution of many organisms (see Frost 1984; Bond & van Wilgen 1996). Yet its role remains contentious and continues to be one of the oldest and most important topics of debate among ecosystems managers (e.g. Scholes & Walker 1993). With a shift in mindset from viewing ecosystems as static to dynamic, ever-changing entities, fire is now regarded as an important agent for stimulating biodiversity. In ecosystems where fires are regular, conservation managers tend to burn under a variety of conditions to maximize the diversity of fire-created habitats (e.g. Walker 1989; Mentis & Bailey 1990), and to allow natural, lightning fires to form the most significant proportion of burns (van Wilgen *et al.* 2000). While some studies show that different experimentally manipulated fire regimes may drive measurable and consistent differences in habitat structure and composition over the short and long term (e.g. Trollope *et al.* 1995; Enslin *et al.* 2000), the ecological significance of these differences to animal communities is less well known.

Fire may have both direct and indirect effects on animal communities. Generally few animals are killed as a direct consequence of being burned in a fire, particularly more mobile species such as birds (Bendell 1974; see Lyon & Marzluff 1985). However, the indirect effects of habitat alteration usually result in more significant changes (see Blake 1982; Dieni & Anderson 1999). Because birds are highly mobile they have the ability to respond rapidly to changes in habitat and food availability (Grinnell 1928). Some studies have shown large changes in bird communities following fires in coniferous forests and shrublands, whereas others have found little response to fire (Lyon & Marzluff 1985; Fraser 1989).

The magnitude of change has been correlated to the severity of the disturbance and the nature of the post-fire habitat (Blake 1982; Fraser 1989). Minor habitat alterations may result primarily in modifications in patterns of habitat use rather than any strong attraction to or avoidance of the modified habitat (Lyon & Marzluff 1985). Bird diversity may even increase in a post-fire landscape, since mild fires may enhance landscape heterogeneity (Bock & Lynch 1970; Raphael *et al.* 1987). Even in more severe burns, bird diversity may be reduced only slightly (Fraser 1989).

Bird community composition has been found to change radically after fires, usually depending on

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*Present address: Kruger National Park, Private Bag X402, Skukuza, 1350 South Africa. E-mail: michaelmills@webmail.co.za the habitat/feeding requirements of each species (Lawrence 1966; Bock & Lynch 1970; Raphael *et al.* 1987; Kreisel & Stein 1999), and is often less predictable than for species communities on unburned sites (Blake 1982). Furthermore, where fires throw habitat into a successional cycle, bird community composition and diversity may change relative to the successional stage of the plant community over a number of years (Raphael *et al.* 1987; Kreisel & Stein 1999). Certain species have been found to be almost entirely restricted to early post-fire habitats, either for feeding (Hutto 1995) or breeding (Frost 1984). This habitat may provide important and unique conditions for these species and be vital for maintaining populations (Hutto 1995).

These studies are almost exclusively from the deciduous forests of North America, where fires have long return periods (Lynham & Stocks 1989; Agee 1994) and post-fire habitats are dramatically different from pre-fire habitats. Few studies have made any quantitative assessment of the effect of fire on bird communities in savanna ecosystems. There are numerous reports of insectivorous birds being attracted to fires to feed opportunistically on fleeing insects or to forage in post-fire habitats (e.g. Barbour 1968; Cooper 1968; Dean 1987). Certain birds also preferentially breed on burnt ground (Frost 1984). However, generally these changes are either short-lived, or pertain to a very specific sub-section of the bird community. This study aims to assess medium-term (two months post-fire) consequences of fires, under different severities of disturbance, to savanna bird communities as a whole.

Research was conducted in the Kruger National Park where the mean fire return period is 4.5 years (van Wilgen *et al.* 2000), much shorter than for other systems where comparable studies of birds have been conducted. For example, fire return periods for temperate deciduous forests are >60 years (Lynham & Stocks 1989; Agee 1994) and is thought to be 10–30 years for fynbos (van Wilgen 1987). Since habitat alteration and floral succession following savanna fires is far less dramatic that in forests or shrublands, and bird community response is typically correlated with magnitude of habitat change, consequences to bird communities are likely to be less striking.

This study aims to test the following predictions:

1. Bird diversity increases after a mild burn, but decreases after a severe burn.
2. Bird community composition changes after a fire, particularly on severe burns, with species feeding in low dense vegetation becoming less common and species feeding on open ground becoming more common.

METHODS

On 4 September 2001 a fire ignited in the southern Kruger National Park (KNP). At 11:00, before the commencement of the fire, conditions were described as 'hazardous'. Relative humidity was 24%, air temperature 32°C and a northeasterly 'Berg wind' was blowing at 16 km/h. Grass biomass was 4128 kg/ha and fuel moisture contents 19.5% on a nearby experimental fire plot which is burned annually, so that biomass contents on the burn were probably in excess of this (A.L.F. Potgieter and South African National Parks, unpubl. data). Further, wind speed increased during the early stages of the fire. Conditions during the early phases of the fire placed it among the most severe savanna fires possible (van Wilgen *et al.* 2003). Owing to varying weather conditions over the burn period (>2 days), fire severity was non-uniform across the landscape (Fig. 1). Since weather conditions were most conducive to a hot fire during the early stages of the burn the general pattern was of a severely burned core with a mildly burned outer zone. In certain areas roads acted as effective fire breaks, creating hard boundaries to the burn.

Two burn treatments were identified at the opposite extremes of burn severity, namely severe towards the centre of the burn, and mild towards the periphery of the burn. Habitat adjacent to the burn acted as a control. Owing to the nested spatial nature of the burn it was impossible to produced matched sites for burn treatments. However, sites were all in the '*Combretum collinum*/*Combretum zeyheri* woodland' landscape type (Gertenbach 1983) with a handful of sites falling on the interface with the 'thickets of the Sabie and Crocodile River' landscape, so that habitat was similar irrespective of burn treatment.

Initially 14 severe, 12 mild and 12 control points were identified. These were spaced at >1 km intervals along access roads and at distances of >300 m but up to 15 km from the burn edge. The geographic coordinates of each site were recorded with a GPS (Garmin eTrex). During the first week of sampling, burn severity at each site was quantified, calculating a severity index to verify the classification based on location within the burn as mild or severe (Appendix 1). This showed a

Fig. 1. The location of the study area within the Kruger National Park (KNP) and a satellite image (Landsat 7; 5 October 2001) of the burn scar/study area (ArcView 1992–2000). Blackness of the image increases with burn severity. Sample sites are indicated with black squares (severe), triangles (mild) or circles (control). Black lines indicate roads.

continuum of burn severity among sites, with the index overlapping between the most heavily burnt mild sites and the most mildly burnt severe sites. To ensure classification as mild or severe, based on location, was justified six intermediate sites were removed from the study, retaining only the 10 severe sites with the highest severity index and the 10 mild sites with the lowest index. Ten control sites were selected, eight of which were matched spatially (<1 km apart) and temporally (always sampled in succession) with mild sites.

Sampling techniques and statistical analyses

Fieldwork was conducted for four weeks, commencing 10 days after the fire had ceased. The 10 mild, severe and control sites were each visited four times, once a week, on each occasion at a different time of the morning. Sampling commenced *c.* 30 min after sunrise and continued until *c.* 10:00. When rain or wind appeared to affect bird activity or detectability, sampling was terminated.

Point counts were conducted for 15 min at each sample point. The time was noted when a species was detected by sight or by sound, the distance between the bird and the sample point estimated and the minimum number of individuals recorded. Birds over-flying the habitat, e.g. raptors, were ignored. Since most records were based on call detection it was difficult to accurately estimate number of individuals. Consequently data on a presence-absence was analysed only. Because density calculation is heavily dependent on accuracy of distance estimation, comparisons based on relative density were used, assuming that detectability was the same for all treatments (Bibby *et al.* 2000).

Mean number of bird species recorded at each sample site ($n = 4$ repetitions) was calculated and used to represent bird diversity for that site. These data were tested for homogeneity and normality using the Levene test of homogeneity

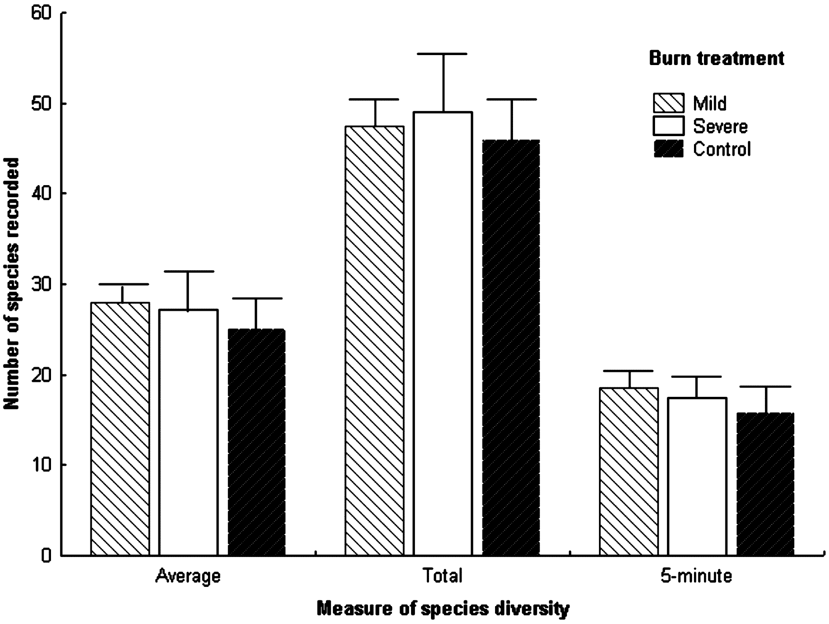

Fig. 2. The number of species recorded at mild, severe and control sites (Mean ± S.D.; $n = 10$ sites per treatment) on average, in total, and on average after 5 min of the 15-min sample period ($n = 4$ rounds). No significant differences existed between the three treatments for any of the measures of species diversity.

($P > 0.05$ assumed to be homoscedastic) and Kolmogorov-Smirnoff goodness of fit test ($P > 0.20$ assumed to be normal) (StatSoft 1993), respectively. Mean and total bird diversity were compared between the three treatments ($n = 10$ sites per treatment) by means of ANOVA tests. Matched mild and control sites were compared by means of the more powerful matched paired *t*-tests (Zar 1999).

The frequency of occurrence of each species by week and over the entire study period was compared between the three treatments ($n = 10$ for each treatment) using both chi-square tests and binomial tests. For the binomial tests the combined frequency of presence ($n = 30$ sites) was used to calculate the expected minimum and maximum number of records per treatment. Chi-square test results are only reported, since they are almost identical to binomial test results.

Each species was classified according to the habitat stratum in which it most often feeds. Categories used were: (1) Upper (branches and leaves of trees), (2) Lower (low thickets, or grass/herb layer) and (3) Ground. The community composition, based on the feeding stratum of species recorded was compared between treatments. Further, ground-dwelling species were divided into two categories; those that prefer feeding on bare/open ground and those that prefer cover. Community composition in terms of predominant diet was also assessed, dividing species into frugivores, granivores and insectivores.

RESULTS

Diversity

Mean ($n = 4$ repetitions) and total bird diversity were strongly correlated (Pearson's correlation: $r = 0.861$, $P < 0.001$, $n = 30$ sites). However, total diversity was highest at severe sites whereas mean diversity was highest at mild sites (Fig. 2) and standard deviation between sites was greatest for the severe treatment. These data suggest that species turnover is higher on severe sites, possibly due to birds having to move more to meet their foraging needs. To test whether this may have affected diversity estimates within the 15-minute sample period, diversity recorded after five minutes was calculated. Here the rank order matched that of mean diversity for the three treatments (Fig. 2). To avoid the confounding effect of different species accumulation patterns for different treatments further investigation of diversity was confined to mean diversity.

Mean diversity was not found to differ between treatments (ANOVA: $F = 1.82$, $P > 0.1$; $n = 10$ sites

Table 1. Species with biased distribution by treatment during any round of sampling, or in total (T), and the number of times recorded for each treatment during that round of sampling. See Appendix 2 for scientific names.

Species	Round	Mild	Severe	Control
Coqui francolin	4	0	0	4
Natal spurfowl	2, T	1, 4	2, 3	7, 9
Red-crested korhaan	4	8	2	4
Ring-necked dove	4	2	6	8
Emerald-spotted dove	1	10	3	8
Little bee-eater	4	5	0	5
African hoopoe	2, 3, T	3, 7, 10	9, 10, 10	1, 2, 5
Common scimitarbill	4	0	4	0
Black-collared barbet	1, 3, T	0, 0, 1	5, 5, 9	0, 1, 3
Bearded woodpecker	Tot	9	10	5
Flappet lark	4	0	5	1
Black-eyed bulbul	2, T	1, 1	6, 6	0, 0
Yellow-bellied eremomela	1	6	0	0
Brubru	3	10	4	8
Brown-crowned tchagra	3	1	1	6
Greater blue-eared starling	1	9	5	2
Scarlet-chested sunbird	3	9	2	4
Blue waxbill	3	5	0	5
Golden-breasted bunting	2	8	7	2

per treatment), although mild sites had the highest diversity (Fig. 2). Matched-pair comparisons between the eight matched mild and control sites, however, showed that mild sites had a higher mean diversity than control sites ($t = 5.24$, $P < 0.002$, $n = 6$).

Community composition

At the species level very few differences were found between the treatments. All species recorded at >7 sites occurred on all three treatments (Appendix 2). The only species that showed a biased distribution across burn treatments and was entirely absent from any one treatment was black-eyed bulbul (*Pycnonotus barbatus*). Other species with biased distributions were black-collared barbet (*Lybius torquatus*), which was more common on severe sites, Natal spurfowl (*Francolinus natalensis*), which was more common on control sites, and bearded woodpecker (*Thripias namaquus*) and African hoopoe (*Upupa africana*), both less common on control sites.

Because combining samples may mask ability to detect subtle differences, differences on a sample-by-sample basis were investigated. Here more biases in species distribution between treatments occurred, although none were consistent over all rounds of sampling (Table 1). Combining all four samples reduced sample variability and provided more complete and robust results for presence/absence comparisons.

Community composition, in terms of the number (Fig. 3; ANOVA: $F < 2.35$, $P > 0.1$, $n = 30$) and proportion (ANOVA: $F < 3.27$, $P > 0.05$, $n = 30$) of species favouring different feeding strata, was not found to differ among treatments for any of the three feeding strata. However, within species that feed on the ground the proportion of species that prefer feeding in cover did differ between the three treatments (Fig. 4; ANOVA: $F = 4.11$, $P < 0.05$, $n = 30$), with 35% on control plots being those that required cover and only 26% and 28% for mild and severe treatments, respectively.

The proportion of birds belonging to each of the three dietary classes showed some differences (Fig. 5). Frugivores made up a significantly larger proportion (ANOVA: $F = 4.28$, $P < 0.025$, $n = 30$; post-hoc LSD test: $P < 0.05$) of birds on severe plots (8.0%) than mild (5.7%) or control plots (5.6%), whereas granivores were relatively more diverse (ANOVA: $F = 10.26$, $P < 0.0005$, $n = 30$; post-hoc LSD test: $P < 0.05$) on mild (14.9%) and control (15.9%) sites than on severe sites (11.9%).

DISCUSSION

Owing to the spatial nature of the burn it was impossible to match mild, severe and control sites. Where it was possible to test for differences in diversity between matched sites, higher diversity on mild compared to control sites was revealed

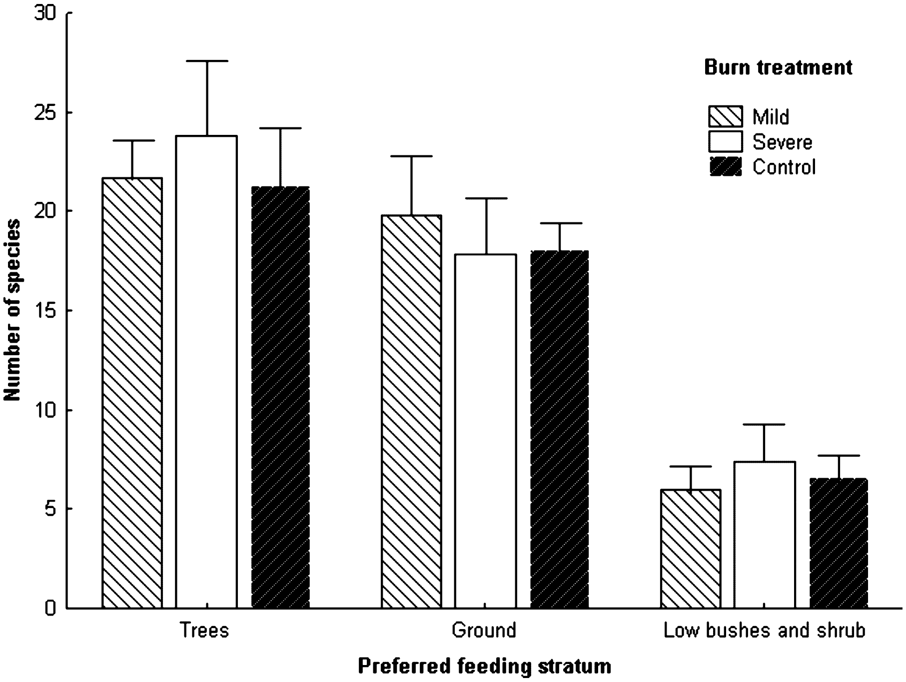

Fig. 3. The number of species recorded per treatment (Mean ± S.D.; *n* = 10 sites per treatment) which prefer feeding in trees, on the ground, or in low bushes and shrubs. No significant differences existed between the three treatments for any of the feeding strata.

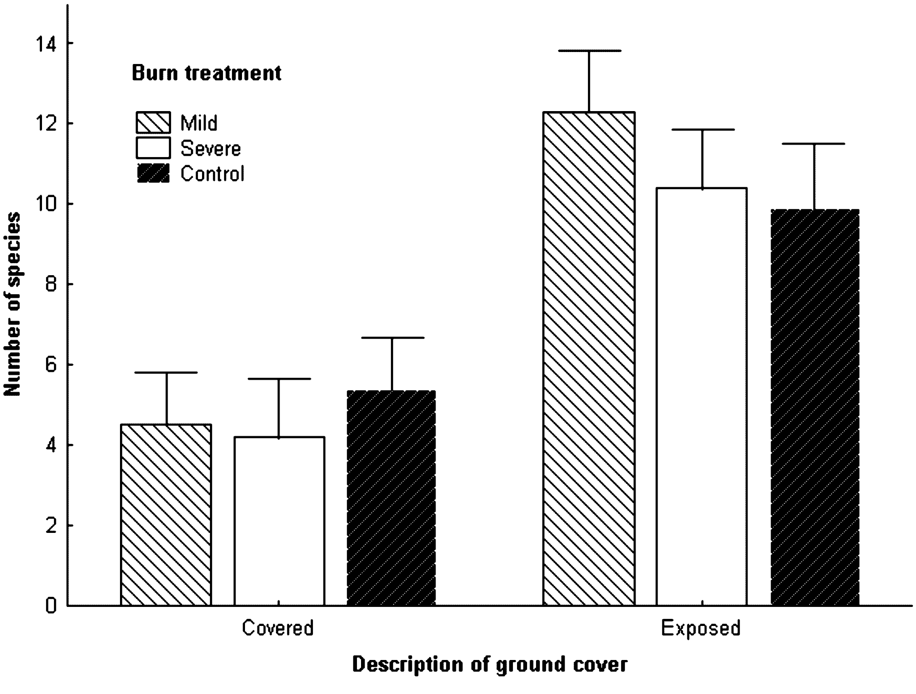

Fig. 4. The number of ground-dwelling species recorded per treatment (Mean ± S.D.; *n* = 10 sites per treatment) which prefer feeding in dense cover, or on open, exposed ground. Control sites yielded significantly more species that preferred feeding in cover, than either of the two treatments.

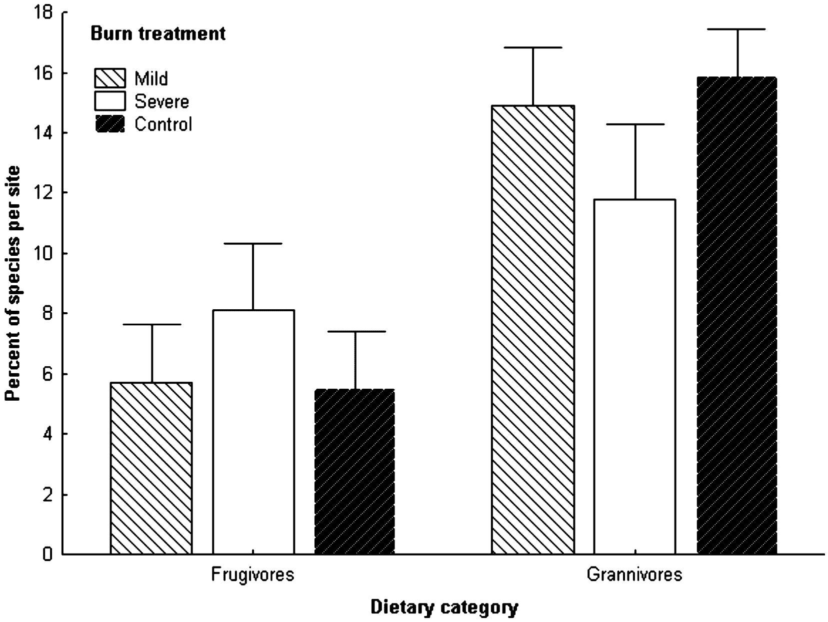

Fig. 5. The proportion of frugivorous and granivorous species recorded per treatment (Mean ± S.D.; *n* = 10 sites per treatment). Frugivores were relatively more abundant on severe sites, whereas granivores were relatively less well represented on severe sites. The remaining species were composed of insectivores.

despite no difference being shown without matching sites. It is clear that geography and time, two of the factors matched by these pairs, are important factors influencing sample diversity, even within a relatively restricted area and times of the day.

Differences in the numbers of species present were, as has been found in most other studies (e.g. Fraser 1989), minor and are best explained by habitat alteration. Fires are hypothesized to increase habitat heterogeneity, consequently making a greater diversity of habitats available to birds (Bock & Lynch 1970; Raphael *et al.* 1987; Hutto 1995). Since mild burns, which provide both bare and sheltered ground, tended to have the highest bird diversity, this study adds weight to the hypothesis.

Unlike in forests (Bock & Lynch 1970; Raphael *et al.* 1987), change in bird community structure following a severe savanna fire seems to be minor. Certain differences detected in this study are best explained by minor differences in pre-fire habitat between the treatments, which could not be controlled for due to the spatial nature of the burn. Most of the severe sites were located in a moister area than the mild and control sites, although still largely in the same landscape, possibly being more suitable for frugivorous species like the black-eyed bulbul and black-collared barbet. It would, however, be interesting to investigate whether certain favoured plant species flower and fruit after fires, as is the case for a number of *Aloe* spp. that occur in the same environment (Frost 1984).

Some changes could be predicted based on habitat alteration. Species that feed on the ground, the stratum where fires alter the structure of savanna vegetation most, were most strongly affected. Those preferring cover, e.g. francolin and spurfowl species, constituted a greater proportion of the bird community on control sites than on either of the treatments. Still, no species was entirely absent from even severely burnt areas. Although differences were not detected for species foraging in dense thickets and shrub, I suspect some subtle changes in density may have occurred. There were also some minor differences when comparing treatments in terms of dietary categories. Species that feed on grass seeds were relatively less common on mild and severe burns, where much of the grass layer had been removed, than on controls.

Perhaps surprisingly, no differences among the insectivorous component of the bird communities were found despite some studies showing strong responses of insect populations to fire (e.g. Gillon 1971). Birds were regularly observed to be feeding on the burn, and since some sites were >10 km from the burn edge, birds recorded were clearly being sustained locally, on the burn. This study suggests that insect communities as a whole, as opposed to certain species of insect, are robust enough to support bird communities, possibly because most savanna insectivores are generalists that feed on a wide range of insect species.

Most of the differences shown in this study are minor and probably have insignificant biological meaning at population levels. Fully controlling for habitat type may have revealed some differences. However, a single savanna fire does not seem to change bird communities in general, even in extreme cases. No species recorded avoided burned habitats completely or was entirely dependent upon fire-created habitats. In deciduous forests, where the habitat is forced into a long-term successional cycle by fire, fire-created habitats are highly distinct and are thought to maintain certain species populations (Hutto 1995). The changes in bird communities following a savanna fire are not only minor, but also are likely to be short-lived. The habitat structure recovers relatively quickly, so that a source-sink situation is unlikely to establish.

Managers of savanna ecosystems can take comfort from these findings. There is clearly little reason to be concerned over the medium-term effects of fires on bird communities, even for the most severe fires. Some savanna species may be dependent on fire-created habitats for breeding (Frost 1984). However, these species tend to be highly mobile, enabling them to locate suitable breeding habitat irrespective of the fire management strategy employed, other than complete fire suppression. Further, the extent of area that burns in savannas has been found to correlate strongly with rainfall (Balfour & Howison 2002) across the employment of a variety of fire management policies (e.g. Brockett *et al.* 2001). While some long-term studies have shown how rigorously applied fire regimes can create structurally distinct habitats (e.g. Trollope *et al.* 1995; Enslin *et al.* 2000), the variety of scenarios applied is likely to range beyond natural variation. A more hands-off approach to fire management is unlikely to have any negative consequences on bird communities and will save time and funds.

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Appendix 1. Burn severity index scores for all sites, ranked from lowest to highest severity. Individual scores are given, in sequence, for remaining grass leaf cover, remaining leaf cover, stem blackening, soil blackening, proportion of vegetation remaining on termitaria, and amount of ash present. Scores were assigned as follows: g*rass and leaf cover*: >75% (1), 75–25% (2), 25–1% (3), 0% (4); s*tem blackening*: none (1), light, at lower levels (2), heavy, at lower levels (3), heavy, throughout (4); s*oil blackening*: none to very black (1–4); *vegetation remaining on termitaria*: 0% (1), 1–50% (2) 50–99% (3), 100% (4); *quantity of ash present*: none (1), few small branches (2), many small branches (3), large quantities, including whole trees (4).

Site	Grass	Score
Mild 2*	1, 1, 2, 1, 1, 1	1.17
Mild 5*	1, 1, 1, 1, –, 2	1.20
Mild 9*	1, 1, 1, 2, 2, 1	1.33
Mild 10*	1, 1, 1, 2, 1, 2	1.33
Mild 4*	1, 1, 3, 1, 2, 1	1.50
Mild 3*	1, 1, 3, 2, –, 1	1.60
Severe 11	3, 2, –, 1, 2, 1	1.80
Mild 7*	2, 3, 2, 2, 3, 1	2.17
Mild 6*	2, –, 2, 2, 4, 1	2.20
Mild 8*	3, 3, 1, 2, 4, 1	2.33
Mild 12*	3, 3, 2, 1, 4, 1	2.33
Severe 3	3, 3, 2, 2, 3, 1	2.33
Severe 5	3, 3, 2, 2, –, 2	2.40
Mild 1	2, 3, 3, 2, 4, 1	2.50
Mild 11	2, 3, 2, 2, 4, 2	2.50
Severe 4	3, 3, –, 3, –, 1	2.50
Severe 2*	4, 3, 2, –, 3, 2	2.80
Severe 1*	4, 3, 2, 2, 4, 3	3.00
Severe 7*	3, 3, 3, 2, 4, –	3.00
Severe 8*	4, 2, 3, –, –, 3	3.00
Severe 14*	3, 3, 3, 2, 3, 4	3.00
Severe 6*	3, 3, 4, 2, 3, 4	3.17
Severe 9*	4, 3, 3, 3, 4, 4	3.50
Severe 10*	4, 3, 3, 3, 4, 4	3.50
Severe 13*	4, 3, 4, 2, 4, 4	3.50
Severe 12*	4, 4, 4, 4, –, 4	4.00

*Sites selected for this study

Appendix 2. The number of mild (M), severe (S) and control (C) sites at which each species was recorded, the total number of sites (T) and the combined frequency of occurrence (F; $n = 30$). Species non-randomly distributed among sites have common names in boldface (chi-square test: $\chi^2 > 7.378$, $P < 0.05$, d.f. = 2). Species recorded <4 times or >26 times are excluded since, statistically, they cannot show biased distributions.

Common name	Scientific name	M	S	C	T	F
Coqui francolin	*Francolinus coqui*	7	6	5	18	0.600
Crested francolin	*Francolinus sephaena*	5	8	8	21	0.700
Natal spurfowl	*Francolinus natalensis*	4	3	9	16	0.533
Swainson's spurfowl	*Francolinus swainsonii*	2	2	5	9	0.300
Laughing dove	*Streptopelia senegalensis*	7	3	7	17	0.567
African green pigeon	*Treron calva*	0	3	1	4	0.133
Brown-headed parrot	*Poicephalus cryptoxanthus*	3	4	5	12	0.400
Grey go-away bird	*Corythaixoides concolor*	9	9	8	26	0.867
African cuckoo	*Cuculus gularis*	2	4	0	6	0.200
Klaas's cuckoo	*Chrysococcyx klaas*	2	1	1	4	0.133
Burchell's coucal	*Centropus burchellii*	2	2	5	9	0.300
Pearl-spotted owlet	*Glaucidium perlatum*	1	6	4	11	0.367
Red-faced mousebird	*Urocolius indicus*	3	1	1	5	0.167
Brown-hooded kingfisher	*Halcyon albiventris*	3	6	4	13	0.433
Striped kingfisher	*Halcyon chelicuti*	5	8	4	17	0.567
Lilac-breasted roller	*Coracias caudata*	10	7	8	25	0.833
African hoopoe	*Upupa africana*	10	10	5	25	0.833
Green woodhoopoe	*Phoeniculus purpureus*	10	7	9	26	0.867
Common scimitarbill	*Rhinopomastus cyanomelas*	8	8	6	22	0.733
Southern ground hornbill	*Bucorvus leadbeateri*	4	3	3	10	0.333
Black-collared barbet	*Lybius torquatus*	1	9	3	13	0.433
Bennett's woodpecker	*Campethera bennettii*	2	1	4	7	0.233
Golden-tailed woodpecker	*Campethera abingoni*	1	5	3	9	0.300
Cardinal woodpecker	*Dendropicos fuscescens*	9	9	7	25	0.833
Bearded woodpecker	*Thripias namaquus*	9	10	5	24	0.800
Flappet lark	*Mirafra rufocinnamomea*	2	4	2	8	0.267
Sabota lark	*Mirafra sabota*	4	2	5	11	0.367
Black cuckooshrike	*Campephaga flava*	2	6	3	11	0.367
African penduline tit	*Anthoscopus caroli*	9	6	9	24	0.800
Black-eyed bulbul	*Pycnonotus barbatus*	1	6	0	7	0.233
Groundscraper thrush	*Psophocichla litsitsirupa*	6	4	2	12	0.400
Yellow-breasted apalis	*Apalis flavida*	2	2	4	8	0.267
Yellow-bellied eremomela	*Eremomela icteropygialis*	7	5	5	17	0.567
Miombo wren-warbler	*Calamonastes stierlingi*	8	5	3	16	0.533
Neddicky	*Cisticola fulvicapillus*	8	10	7	25	0.833
Tawny-flanked prinia	*Prinia subflava*	0	4	1	5	0.167
Southern black flycatcher	*Melaenornis pammelaina*	6	8	3	17	0.567
Pale flycatcher	*Bradornis pallidus*	6	3	5	14	0.467
African pipit	*Anthus cinnamomeus*	1	3	0	4	0.133
Bushveld pipit	*Anthus caffer*	9	7	4	20	0.667
Magpie shrike	*Corvinella melanoleuca*	10	5	7	22	0.733
Southern boubou	*Laniarius ferrugineus*	1	5	1	7	0.233
Black-backed puffback	*Dryoscopus cubla*	1	2	2	5	0.167
Brown-crowned tchagra	*Tchagra australis*	4	5	9	18	0.600
Orange-breasted bushshrike	*Telophorus sulfureopectus*	1	3	6	10	0.333
Grey-headed bushshrike	*Malaconotus blanchoti*	4	5	4	13	0.433
White helmetshrike	*Prionops plumatus*	2	5	4	11	0.367
Southern white-crowned shrike	*Eurocephalus anguitimens*	2	2	1	5	0.167
Violet-backed starling	*Cinnyricinclus leucogaster*	1	4	1	6	0.200
Burchell's starling	*Lamprotornis australis*	6	4	8	18	0.600
Cape glossy-starling	*Lamprotornis nitens*	10	8	8	26	0.867

Continued on p. 11

Appendix 2 (*continued*)

Common name	Scientific name	M	S	C	T	F
Greater blue-eared starling	*Lamprotornis chalybaeus*	10	9	7	26	0.867
Marico sunbird	*Nectarinia mariquensis*	4	4	5	13	0.433
White-bellied sunbird	*Nectarinia talatala*	6	9	4	19	0.633
Scarlet-chested sunbird	*Nectarinia senegalensis*	10	6	9	25	0.833
Southern grey-headed sparrow	*Passer griseus*	4	3	4	11	0.367
Red-headed weaver	*Anaplectes rubriceps*	0	1	4	5	0.167
Green-winged pytilia	*Pytilia melba*	3	0	3	6	0.200
Jameson's firefinch	*Lagonosticta rhodopareia*	2	0	3	5	0.167
Blue waxbill	*Uraeginthus angolensis*	9	5	6	20	0.667

Species recorded at >26 sites: red-crested korhaan *Eupodotis ruficrista,* ring-necked dove *Streptopelia capicola,* emerald-spotted dove *Turtur chalcospilos,* african grey hornbill *Tockus nasutus,* southern yellow-billed hornbill *Tockus leucomelas,* crested barbet *Trachyphonus vaillantii,* fork-tailed drongo *Dicrurus adsimilis,* eastern black-headed oriole *Oriolus larvatus,* southern black tit *Parus niger,* arrow-marked babbler *Turdoides jardineii,* white-browed scrub-robin *Cercotrichas leucophrys,* long-billed crombec *Sylvietta rufescens,* rattling cisticola *Cisticola chinianus,* chinspot batis *Batis molitor,* brubru *Nilaus afer,* black-crowned tchagra *Tchagra senegala,* yellow-throated petronia *Petronia superciliaris,* yellow-fronted canary *Serinus mozambicus,* golden-breasted bunting *Emberiza flaviventris.*