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Title: Effects of a short fire-return-interval on resources and assemblage structure of birds in a tropical savanna

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Running title: Short fire-return-intervals and birds

Short fire-return-intervals and birds
Summary

Fire frequency is a key land management issue, particularly in tropical savannas where fire is widely used and fire recurrence times are often short. We used an extended BACI (Before-After-Control-Impact) design to examine the impacts of repeated wet-season burning for weed control on bird assemblages in a tropical savanna in north Queensland, Australia. Experimentally replicated fire treatments (unburnt, singularly bunt, twice burnt), in two habitats (riparian and adjacent open woodland), were surveyed over three years (1 year before 2\textsuperscript{nd} burn, 1 year post 2\textsuperscript{nd} burn, 2 years post 2\textsuperscript{nd} burn) to examine responses of birds to a rapid recurrence of fire. Following the second burn, species richness and overall bird abundance were lower in the twice burnt sites than either the unburnt or singularly burnt sites. Feeding group composition varied across year of survey, but within each year, feeding guilds grouped according to fire treatment. In particular, abundance of frugivores and insectivores was lower in twice burnt sites, probably due to the decline of a native shrub that produces fleshy fruits, *Carissa ovata*. Although broader climatic variability may ultimately determine overall bird assemblages, our results show that a short fire-return-interval will substantially influence bird responses at a local scale. Considering that fire is frequently used as a land management tool, our results emphasise the importance of determining appropriate fire-free intervals.

Key words: alien plant species, BACI, birds, feeding guild, fire, fire frequency, tropical savanna, riparian habitat, rubber vine.

Short fire-return-intervals and birds
Introduction

Fire is a widespread influence on ecosystems, and is one of the most powerful tools available for extensive management of vegetation. The human-mediated use of fire has important ramifications for the environment, and fire frequency has emerged as a key attribute of fire regime (Williams et al. 2003), particularly because it is often manipulated for land management practices. Australia is the most fire-prone continent on Earth (Bowman 2003), and human-mediated fire has influenced the Australian landscapes for at least 5,000 years (Johnson 2006). In particular, the extensive tracts of tropical savannas in northern Australia are subjected to a high prevalence of fire. The characteristic seasonality of tropical savannas, with a distinct wet season followed by an extended dry season, ensures that low-intensity fires are frequent (Russell-Smith et al. 1997). Indeed, most savanna habitat in Australia receives fire at least once every five to ten years (Lacey et al. 1982) and large areas are burnt annually (Russell-Smith et al. 1997).

Although tropical savannas and associated biota are often considered resilient to the effects of fire (Andersen et al. 2005), certain elements, such as riparian or monsoon forests, may be adversely affected by high fire frequencies (Andersen et al. 2005; Russell-Smith and Bowman 1992) and there is evidence that inappropriate fire regimes are linked to the decline of several bird and mammal species (Franklin 1999; Franklin et al. 2005; Pardon et al. 2003). Thus, understanding the ecological impacts of frequent fire is critical for land managers, particularly in landscapes where fire is often used for landscape management.

A practical land management application of fire in northern Australia is for the control of invasive alien plants (Emery and Gross 2005; Grice 1997). Rubber vine
(Cryptostegia grandiflora) is one of Australia’s most significant invasive species, due to its deleterious affect on native ecosystems (Commonwealth of Australia 1999). This globally distributed woody weed smothers native vegetation and threatens native biodiversity values of tropical savannas, especially within the riparian zones (Tomley 1998; Valentine 2006; Valentine et al. 2007a). Fire can reduce rubber vine survival, and therefore burning may be a useful tool to control rubber vine infestations (Bebawi and Campbell 2000, 2002; Grice 1997). Burning an area twice, with a short fire-return-interval is often required to control woody weeds (Vitelli and Pitt 2006), including rubber vine (Radford et al. 2008). However, little is known about the potential impacts of repeated management burning on native fauna over short time frames.

The effects of fire frequency on communities is often ascertained by comparing sites with different fire histories (e.g. high versus low: Andersen et al. 2005; Govender et al. 2006; Parr et al. 2004; Woinarski 1990; Woinarski et al. 1999) and generating correlative evidence on the biotic responses. Here, we use an experimental approach to understand the mechanisms of effects of regular fires in a tropical savanna. Using an experimental framework in which some sites were burnt, while others were not, to measure the effects of fire (Radford et al. 2008; Valentine et al. 2007b), we then reburned some of the burnt plots as soon as possible following the first fire (i.e. after two years) to measure the effects of a short fire-return-interval on bird assemblages. This design allows us to examine the impact of a repeat burn using a before-after/control-impact (BACI; Stewart-Oaten et al. 1986) design, a technique few fire studies allow (but see Smucker et al. 2005). Here, we experimentally examine the response of bird assemblages to a short fire-return-interval in grazed tropical savannas.
Methods

Study Site and Experimental Design

The study took place in open eucalypt woodlands of the Einasleigh Uplands bioregion (Sattler and Williams 1999), 110 km south of Townsville in north-eastern Queensland, Australia. The experimental design was initially established by the Commonwealth Scientific and Industry Research Organisation – Sustainable Ecosystems (CSIRO-SE) and Tropical Savannas Cooperative Research Centre (TS-CRC) in 1999 for investigating the effectiveness of fire regimes to control rubber vine (Radford et al. 2008). Sites were located along three seasonally dry watercourses: Bend Creek (20°16′07″S, 146°37′48″E), One Mile Creek (20°14′10″S, 146°40′35″E) and Cornishman Creek (20°12′18″S, 146°27′15″E), all tributaries of the Burdekin River on commercial lease-hold grazing properties (Figure 1). Along each watercourse approximately three km was divided into landscape size experimental plots that were randomly assigned to a prescribed burning treatment and encompassed both riparian and adjacent non-riparian woodland habitat (see Radford et al. 2008 for habitat description; Valentine et al. 2007b), henceforth referred to as woodland. Rubber vine occurred in low to moderate infestations in both habitats, but was more prolific in riparian habitat. Each plot was approximately 20 ha in size (encompassing both habitats) and separated from other plots by double fire breaks, spaced at least 50 m apart. On each creek, the same experimental treatments were randomly imposed upon plots and included: i) an unburnt control plot; ii) a plot that was burnt once, in December 1999, and iii) a twice-burnt plot, initial fire imposed December 1999 and second fire imposed December 2001 (Fig. 1). Fire treatments examined in this paper were imposed during the early tropical wet season, following the first substantial rains. In lands used for cattle Short fire-return-intervals and birds
growing, managers often burn at the beginning of the wet season to maximize the pasture growth response to fire (Crowley and Garnett 2000). Previous research has also indicated that wet season fires have less effect on bird assemblages than mid-dry season fires (Valentine et al. 2007b). A detailed account of the implementation of prescribed burning action undertaken is provided in Radford et al. (2008). Here, site refers to either the woodland or riparian habitat of each fire treatment, providing a total of six sites along each water course (one site per habitat of each fire treatment).

Sampling Strategy

Data were collected during the tropical wet season between January and March 2001, 2003 & 2004. Sampling at these times allowed data collection one year after the first burn, one year after the second burn, and two years after the second burn (Table 1). Plots were surveyed in random order within each creek. Bird assemblages were surveyed using two, thirty-minute timed surveys along 250 m line transects running through the middle of each site (habitat/treatment replicate) using the methodology described in Valentine et al. (2007b). The abundance of all individual birds observed per transect within a site was averaged. The number of species observed was summed to estimate total species richness for each site. Data on vegetation variables were collected using four large quadrats (20 m x 10 m) in each site. In each quadrat, the number and species of trees and shrubs were recorded. The proportion of rubber vine cover in the understorey was estimated to the nearest 10% in each quadrat. In the last two years of surveys, several plants were observed fruiting, particularly the commonly occurring native shrub currant bush (*Carissa ovata*). Several birds were observed feeding upon this shrub (e.g. noisy friarbirds *Philemon corniculatus* and great bowerbirds *Chlamydera nuchalis*). To examine the potential change in food resources Short fire-return-intervals and birds
provided by this shrub, we estimated the area (m²) of currant bush within each quadrat in the last two years of surveys. All vegetation data were averaged among the four quadrats to provide an estimate of average vegetation structure within each site.

**Analysis**

*Species richness and bird abundance*

The experiment had a randomized block design and provided the opportunity to use a Before-After-Control-Impact (BACI; Stewart-Oaten *et al.* 1986) approach to examine the impact of the second fire on mean bird abundance and species richness. The BACI ANOVA (Analysis of Variance; conducted in SPSS, version 12) included time (1 year before 2nd fire, 1 year after 2nd fire and 2 years after 2nd fire), fire treatment (unburnt, singularly burnt and twice burnt) and habitat (riparian and woodland) as main effects terms, and creek as a blocking factor. Interaction terms for time*fire treatment and time*fire treatment*habitat were also included to examine the impact of the second burn. If a significant interaction was detected, indicating a significant impact of the second burn, post-hoc ANOVAs were used to further explore sources of variation among factors within each year of survey. Post-hoc ANOVA used fire treatment and habitat as main effects terms (including an interaction term) and creek as a blocking factor. To identify species that may be driving patterns within each year of survey, the responses of 24 commonly occurring bird species (i.e. species observed in at least 20 of the possible 54 time*fire treatment*habitat sites) were examined using ANOVA.

*Feeding group assemblage*

Bird species were assigned to one of five feeding groups (carnivore, granivore, frugivore, insectivore and nectarivore) based on their dominant source of food, using

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dietary information from the literature (Catterall and Woinarski 2003; Higgins 1999; Higgins and Davies 1996; Higgins and Peter 2002; Higgins et al. 2005; Higgins et al. 2001; Marchant and Higgins 1990a, b 1993). Feeding group assemblage was examined by comparing the average number of carnivores, granivores, frugivores, insectivores and nectarivores at each site using a MANOVA (SPSS, version 12) with the BACI design (described above). A Canonical Discriminant Analysis (CDA) was used to interpret significant interaction terms and to examine the influence of time and fire treatments on feeding group assemblage. Abundance of each feeding group was subsequently examined independently, using the BACI ANOVA described above (with time, fire treatment, habitat and creek as factors). If a significant time*fire treatment interaction term was detected, post-hoc ANOVA was used to further explore sources of variation within each year of survey (described above).

Vegetation structure and birds

Vegetation structure was examined by comparing the average number of trees, shrubs and the proportion of rubber vine per quadrat in each site using a MANOVA (SPSS, version 12), with the BACI design (described above). The area of currant bush was compared each year surveyed using ANOVA (described above). Species richness, bird abundance and feeding group abundance and the abundance of species that responded significantly to the second fire were compared with vegetation variables within each year of survey using Pearson’s correlation coefficients.

Data transformations

Data on rubber vine percentage were adjusted by arcsine square-root transformation (Zar 1999). Species richness, abundance counts (birds, feeding groups,
trees and shrubs) and amount of currant bush (m$^2$) were examined for normality and heteroscedasticity using box plots, Q-Q plots and residual plots. Multivariate linearity was determined by examination of scatterplots of variables used in MANOVA. Species richness and abundance of birds, trees and shrubs, and the amount of currant bush were square-root transformed and individual species abundances and feeding group abundances were log$_{10}$(x + 1) transformed to meet assumptions of ANOVA and Pearson’s correlation. To aid interpretation, graphs depict the untransformed data.

8 Results

9 Species richness and bird abundance

A total of 67 bird species were identified during the three years of surveys. An additional three species were observed once each during surveys but not positively identified. The unknown species were included in abundance analyses but were excluded from species richness and feeding group analyses. BACI analysis on the square-root transformed species richness detected a significant interaction among time, fire treatment and habitat (Table 2; Fig 2.). This interaction showed that the number of species in the twice-burnt sites was lower than the single-burnt or unburnt sites one year after fire in both habitat types and two years after the second fire in the riparian habitat, but not the woodland habitat. A significant difference in species richness was also observed among creeks. Cornishman creek contained the least number of species compared to either Bend Creek or One Mile Creek (species richness mean [±95%CI]: Cornishman Creek = 17.05 [1.33]; Bend Creek = 21.94 [2.35]; One Mile Creek = 20.66 [2.09]).
One year before the second fire, post-hoc ANOVA revealed similar number of species in all fire treatments (Fig. 2), although species richness differed among creeks (ANOVA: $F_{2,10} = 7.722, P = 0.009$). One year after the second fire, species richness in the twice burnt sites was lower than in unburnt and singularly burnt sites (ANOVA: $F_{2,10} = 11.459, P = 0.003$; Fig. 2) and varied among creeks (ANOVA: $F_{2,10} = 5.032, P = 0.031$). Two years following the second fire, species richness was still lower in twice burnt sites compared to unburnt and singularly burnt sites (ANOVA: $F_{2,10} = 9.127, P = 0.006$; Fig. 2). Species richness was also lower in the woodland habitat than in the riparian habitat (ANOVA: $F_{1,10} = 9.444, P = 0.010$). It is in this year that the difference between habitats became apparent with regards to fire treatment. The single burnt sites in riparian habitat had higher species richness compared to the twice burnt sites, while the single burnt sites in woodland habitat did not differ to the twice burnt sites (Fig 2).

At least six species, including great bowerbird (*Chlamydera nuchalis*), red-backed fairy-wren (*Malurus melanocephalus*), yellow honeyeater (*Lichenostomus flavus*), Lewin’s honeyeater (*Meliphaga lewinii*), noisy friarbird (*Philemon corniculatus*) and rainbow lorikeet (*Trichoglossus haematodus*), that were regularly observed in the unburnt and singularly burnt sites were absent from the twice-burnt sites in at least one of the years following the second fire.

We detected a significant interaction between time and fire treatment on the square-root transformed bird abundance (ANOVA: $F_{4,34} = 7.310, P < 0.001$; Fig. 3, Table 2), such that the total abundance of birds decreased following the second fire. Post-hoc ANOVA conducted in each year of survey initially showed higher abundance of birds in the burnt sites compared to the unburnt sites (ANOVA: $F_{2,10} = 6.224, P = 0.018$; Fig. 3). However, one year following the second fire, abundances of birds in the
twice burnt sites were lower than the unburnt (ANOVA: $F_{2,10} = 5.463, P = 0.025$; Fig. 3) and by two years following the second fire, abundances were lower than both unburnt and singularly burnt sites (ANOVA: $F_{2,10} = 8.422, P = 0.007$; Fig. 3). In the final year of survey, abundances of birds differed between habitat types, with lower abundances of birds observed in the woodland habitat (ANOVA: $F_{1,10} = 13.988, P = 0.004$).

Responses of individual species

A number of species ($n= 16$) showed differences in abundances among fire treatments and/or between habitats (Appendix 1). Before the second fire, striated pardalotes ($Pardalotus striatus$) were more abundant in burnt sites. An interaction was detected between habitat and fire treatment for pied butcherbirds ($Cracticus nigrogularis$) and black-faced cuckoo-shrikes ($Coracina novaehollandiae$), graphical interpretation indicated that birds were more abundant in the woodland habitat of burnt sites. In contrast, the red-backed fairy-wren ($Malurus melanocephalus$) showed a trend for lower abundances in the burnt sites. In the two years following the second fire, a few species, including the frugivore mistletoebird ($Dicaeum hirundinaceum$), the granivore red-winged parrot ($Aprosmictus erythropterus$), the insectivore white-throated honeyeater ($Melithreptus albogularis$), and the nectarivores noisy friarbird ($Philemon corniculatus$) and rainbow lorikeet ($Trichoglossus haematodus$), had lower abundance in the twice burnt sites compared to either the singularly burnt sites, unburnt sites, or both (Appendix 1). In addition, an interaction between habitat and fire treatment was detected for great bowerbirds ($Chlamydera nuchalis$), graphical interpretation indicated that abundance of this species was lower in the woodland habitat of singularly burnt sites and both habitats of the twice-burnt sites, compared to unburnt sites or the riparian habitat of singularly burnt sites. Five bird species were detected at some point during
the surveys with higher abundances in the riparian habitat, and five different bird
species were detected with higher abundances in the woodland habitat (Appendix 1).

**Feeding group assemblage**

We detected an interaction between time and fire treatment in the abundances of
feeding groups (MANOVA Wilks’ Lambda: $F_{20,100.5} = 1.857, P = 0.024$) and significant
variation in feeding group abundance among creeks (MANOVA Wilks’ Lambda: $F_{10,60}$
$= 13.674, P < 0.001$) and habitats (MANOVA Wilks’ Lambda: $F_{5,30} = 2.568, P =$
$0.048$). A Canonical Discriminant Analysis (CDA) was used to examine the influence
of the time*fire treatment interaction on feeding group assemblage (Fig. 4). Time
separated along the first discriminate axis while fire treatments separated along the
second (Fig. 4). Before the second fire, burnt sites grouped together and were
influenced by the abundance of granivores and carnivores at these sites (Fig. 4b and 4e).
However, following the second fire, unburnt sites grouped away from the twice burnt
sites, a grouping that was influenced by the abundance of frugivores, insectivores and
nectarivores. During this year of sample the singularly burnt sites were scattered
throughout, reflecting the variability in the abundance of the different feeding groups at
these sites (Fig. 4a and 4e). The pattern of separation between unburnt and twice burnt
sites continued two years following the second fire, with unburnt sites grouping together
influenced by the abundance of frugivores. Singularly burnt sites clustered between the
two fire treatments (Fig. 4c).

The abundance of each feeding group was examined individually, and we
detected a significant interaction in the abundance of frugivores between time and fire
treatment (ANOVA: $F_{4,34} = 3.299, P = 0.022$; Fig. 5a). Frugivore abundance was
consistently low during the first year of survey, before the second fire. However,
following the second fire, frugivores were least abundant in twice burnt sites (Fig. 5a).

The abundance of frugivores also varied among creeks (ANOVA: $F_{2,34} = 3.546$, $P = 0.040$), with lower abundance of frugivores observed at One Mile creek compared to Bend creek (Tukey HSD: $P = 0.035$; untransformed means [± 95%CI]: One Mile = 0.9 [0.4], Bend = 1.8[0.8]). A significant interaction between time and fire treatment was also detected for insectivores (ANOVA: $F_{4,34} = 3.010$, $P = 0.031$; Fig. 5c) which showed similar patterns to insectivores. Nectarivore abundance was also highest in the riparian habitat (ANOVA: $F_{1,34} = 11.173$, $P = 0.002$; untransformed means [95%CI]: riparian = 7.2 [1.1], woodland = 5.2 [1.1]) and varied among creeks (ANOVA: $F_{2,34} = 4.967$, $P = 0.013$). Nectarivores were observed in higher abundances at Bend creek compared with Cornishman creek (Tukey HSD: $P = 0.014$; untransformed means [95%CI]: Bend = 7.8 [1.8], Cornishman = 5.2 [1.2]). There was no significant interaction term for either granivores or carnivores (Fig. 5d and 5e). Granivore abundance showed a marked decline over time (ANOVA: $F_{2,34} = 18.218$, $P < 0.001$) and was particularly low in the final year of surveys (Fig. 5d). The abundance of carnivores similarly varied over time (ANOVA: $F_{2,34} = 11.175$, $P < 0.001$) and was greatest in the first year of surveys (Fig. 5e). There was also some variation in the abundance of carnivores among creeks (ANOVA: $F_{2,34} = 3.378$, $P = 0.046$), with the abundance of carnivores higher at One Mile creek compared to Bend creek (TukeyHSD: $P = 0.037$; untransformed means [95%CI]: One Mile = 3.4 [0.6], Bend = 2.6 [0.9]).
Vegetation structure

We detected a significant difference in vegetation structure between habitats (MANOVA Wilks’ Lambda: $F_{3,32} = 51.589, P < 0.001$), and among treatments (MANOVA Wilks’ Lambda: $F_{6,64} = 4.499, P < 0.001$) and creeks (MANOVA Wilks’ Lambda: $F_{6,64} = 2.756, P = 0.019$), but no significant interaction between time and fire treatment. The abundance of trees was higher in the riparian habitat and varied among creeks (Table 3). Shrub abundance was lower in the burnt sites compared to the unburnt sites and also varied among creeks (Table 3). The proportion of rubber vine was lower in the woodland habitat and in the burnt sites (Table 3).

During the first year of survey, few plants were observed fruiting. However, in the remaining two sample years, surveys coincided with fruiting plants, one of the most common of which was the low native shrub currant bush (*Carissa ovata*). One year following the second burn, the amount of currant bush (*Carissa ovata*) was lower in the twice burnt sites compared to the unburnt (ANOVA: $F_{2,10} = 7.472, P = 0.010$; mean square-root m$^2$ per quadrat ± 95%CI: unburnt = 2.5 ± 1.0, singularly burnt = 1.4 ± 1.3, twice burnt = 0.5 ± 0.5) and varied among creeks (ANOVA: $F_{2,10} = 9.596, P = 0.005$). Similarly, the amount of currant bush was lower in the twice burnt sites two years following the second fire (ANOVA: $F_{2,10} = 5.720, P = 0.022$; mean square-root m$^2$ per quadrat [95%CI]: unburnt = 2.9[1.3], singularly burnt = 1.1 [1.0], twice burnt = 0.7 [0.5]).

Birds and vegetation

Prior to the second fire, the abundances of both frugivores and insectivores were negatively correlated with the number of trees and the proportion of rubber vine in the understorey (Table 4), indicating that frugivores and insectivores were associated with Short fire-return-intervals and birds.
woodland habitat, and insectivores may also have selected burnt sites. However, following the second fire, the abundance of birds, frugivores and nectarivores, as well as species richness were positively associated with trees, rubber vine or currant bush (Table 4), indicating that several birds were associated with unburnt or riparian habitat. In particular, a positive correlative relationship was observed between the abundance of frugivores and the amount of currant bush (Table 4, Fig. 6). Several great bowerbirds (Chlamydera nuchalis) were observed feeding on the berries of currant bush, and the abundances of great bowerbirds and mistletoebirds (Dicaeum hirundinaceum) were positively associated with the amount of currant bush (Table 4). The abundance of nectarivores, including noisy friarbirds (Philemon corniculatus), was also strongly associated with currant bush.

Discussion

The impact of a short fire-return-interval

Repeat fires, occurring within short-time frames, substantially altered bird assemblages in grazed tropical savannas. Prior to the second fire, both singularly burnt fire treatments contained higher overall abundance of birds. Increased abundance or species richness of birds in recently burnt sites has been observed in a variety of studies (Braithwaite and Estbergs 1987; Hutto 1995; Valentine et al. 2007b; Woinarski 1990), and is attributed to a temporary change in resource availability that benefits some species (Brawn et al. 2001; Saab and Powell 2005; Woinarski and Recher 1997).

In contrast, following the second fire, there was a reduction in species richness, total bird abundance, the abundance of some feeding groups, and particular species. This suggests that burning for a second time within two years of a fire alters habitat in a
manner unfavourable to many birds. As the responses of birds to disturbances are often associated with changes in the availability of resources and vegetation structure (Brawn et al. 2001; Davis et al. 2000; Kutt and Woinarski 2006; Woinarski and Recher 1997), it is likely that the second fire reduced the quantity or quality of resources important for some species. Interestingly, the abundance of birds and species richness tended to increase in the unburnt sites the first year after the 2nd fire. Given the proximity of the fire treatments, it is possible that birds were moving into the unburnt sites to forage, although we have no supporting data. Dissimilarity in bird assemblages between areas with high versus low burning frequency has been observed in oak savannas (Davis et al. 2000) and Brazilian rainforests (Barlow and Peres 2004; Barlow et al. 2006) where differences were attributed to changes in vegetation structure and resource availability. Similarly, previous studies in tropical savannas have identified fire frequency as an important influence on bird feeding groups (Woinarski 1990; Woinarski et al. 1999).

**Changes in food availability – the response of bird feeding groups**

Feeding group assemblages varied among year of survey and fire treatments. In particular, the assemblages of feeding groups were strongly separated by year of survey, and probably reflect differences in conditions (e.g. mean annual rainfall). Year-to-year variability in bird assemblages is fairly common in tropical savannas (Woinarski and Tidemann 1991) and is best explained by changes in resource availability that relate to the extremes of within-year seasonality (Woinarski and Tidemann 1991), or differences in the wet season characteristics among years (Taylor and Tulloch 1985). However, within each year of survey, fire treatments showed distinct feeding group assemblages. Although burnt sites initially grouped together, in the two years following the second
fire, feeding group assemblages of twice burnt sites diverged increasingly from unburnt
and singularly burnt sites.

Frugivore abundance was initially low in all fire treatments when few plants
were observed fruiting. However, in later years, fruiting shrubs coincided with bird
surveys and the abundance of frugivores was lowest in the twice burnt sites. Frequent
fires can reduce the abundance and structure of understorey shrubs (Fox and Fox 1986)
and burning may also reduce the number of fruiting species (Sanaiotti and Magnusson
1995) or fruit production (Setterfield 1997) in the post-fire environment. Berries of the
currant bush provide a food resource for frugivores like great bowerbirds (*Chlamydera
nuchalis*), and may also provide foraging opportunities or shelter for other species. As
the amount of currant bush was reduced in twice burnt sites, frugivores may have been
affected by the loss of food resources.

Following the second fire, the overall abundance of insectivores was lower in
twice burnt sites. This pattern was probably driven by reduced abundance of white-
throated honeyeaters (*Melithreptus albogularis*). Frequent low intensity-fires can
reduce invertebrate abundance and species richness in subtropical eucalypt forests
(York 1999, 2000). Although tropical savanna arthropod communities tend to be
resilient to fire (Andersen *et al.* 2005; Parr *et al.* 2004), burning does disadvantage
certain arthropods and alter the overall composition of arthropod communities
(Andersen and Muller 2000; Parr *et al.* 2004), potentially to the disadvantage of
insectivorous species. Further, the change in invertebrate communities caused by
repeated burning may be amplified in environments with additional disturbances, such
as introduced species (Valentine *et al.* 2007a) and grazing (Abensperg-Traun *et al.*
1996). A reduction in preferred arthropod abundance may also influence the response

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of some nectarivores that include arthropods in their diet, such as noisy friarbirds (Philemon corniculatus) (Higgins et al. 2001).

Changes in habitat structure

Our study examined the effects of a single short fire-return-interval, and both singularly and twice burnt sites had a lower vegetation complexity than unburnt sites. In addition, the twice burnt sites contained lower amounts of currant bush than the unburnt sites. Given that currant bush creates a dense shrubby understorey, the removal of this plant may reduce both available shelter and foraging opportunities for birds. In other parts of northern Australia, where short fire-return-intervals result in frequent fires in tropical savannas, the seedling establishment in shrub and tree species may be reduced (Andersen et al. 2005). Frequent fires may also reduce plant species richness (Fensham 1990), particularly of obligate seeding species (Russell-Smith et al. 1998). Further, frequent fire in tropical savannas typically leads to reduced vegetation complexity (Bowman et al. 1988; Christensen et al. 1981) and may disadvantage bird species by removing important nesting or foraging resources (Artman et al. 2001).

Habitat differences between creeks are likely to be driving some of the differences in bird assemblages and abundance that were also observed. The creeks examined in our study were typical of the region, but they did display variability in terms of the width of creek line, distance to the Burdekin River and underlying cattle management practices. Bend creek had the largest creek width, while One Mile creek had the smallest. Both Bend and One Mile creeks were located on one grazing station, while Cornishman creek was located on an adjacent property and may have experienced different cattle grazing management practices.

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Conservation and Management Implications

Given the deleterious impacts of rubber vine on the biodiversity values of native ecosystems (Commonwealth of Australia 1999), its control is of vital importance for land managers. Fire is by far the most economical and effective tool for reducing rubber vine infestations (Grice 1997; Tomley 1998). In our study, burning reduced the amount of rubber vine, and elements of rubber vine vegetation (e.g. rubber vine towers) were visibly less in the twice burnt sites. Prescribed burning to control rubber vine during the early wet season is more effective at reducing rubber vine infestation when the fire is repeated within a short-time frame (Radford et al. 2008). However, our results show that two fires in quick succession may have unwanted effects on native birds within the short-term. The longer-term consequences of burning for weed control using a short fire-return interval on birds needs to be examined. The long-term bird assemblage structure is likely to be shaped by subsequent management actions at these sites (including no follow-up management).

Fire is clearly a strong influence on the structure of bird assemblages (Woinarski & Recher, Valentine 2007, Davis et al 2000). In this study, we showed that re-burning riparian habitat within two years of a fire reduces species richness and the abundance of some bird species. Frequent burning in tropical savannas can simplify habitat structure (Bowman et al. 1988), especially in the riparian zones which tend to have high structural complexity, reducing resources for birds. In areas subject to a range of disturbances, including grazing and introduced plant species, frequent fires may further simplify habitats. The current burning paradigm is variable throughout tropical savannas, but there is a tendency for managers to set fires in the mid-dry or wet seasons to reduce the likelihood of destructive, late dry season fire (Crowley and Garnett 2000; Short fire-return-intervals and birds
We recommend that managers exercise caution with respect to the frequency with which they burn. Specifically, burning within a short fire-free-interval will have negative effects on bird diversity in riparian habitats of tropical savannas. Further work is needed to determine appropriate fire-free intervals that meet both land management and conservation objectives.

Acknowledgements

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References


Short fire-return-intervals and birds
Valentine, Schwarzkopf and Johnson


Short fire-return-intervals and birds
Short fire-return-intervals and birds


R.G. & F.J. Richardson, Melbourne, Australia.


Short fire-return-intervals and birds


Short fire-return-intervals and birds
Table 1. The experimental design showing the recent fire history (year since last fire) of sites for each combination of fire treatment and time. The columns highlighted indicate the years that bird surveys were conducted.

<table>
<thead>
<tr>
<th>Fire treatment</th>
<th>Year of survey (as of January)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2001 (before 2&lt;sup&gt;nd&lt;/sup&gt; fire)</td>
</tr>
<tr>
<td>Unburnt</td>
<td>≥ 10 years</td>
</tr>
<tr>
<td>Singly burnt</td>
<td>1 year</td>
</tr>
<tr>
<td>Twice burnt</td>
<td>1 year</td>
</tr>
</tbody>
</table>
Table 2. ANOVA F-values for species richness and abundance of birds using BACI. Significant values are in bold (* $P < 0.5$, ** $P < 0.01$, *** $P < 0.001$).

<table>
<thead>
<tr>
<th></th>
<th>Time df=2,34</th>
<th>Fire treatment df=2,34</th>
<th>Habitat df=1,34</th>
<th>Creek df=2,34</th>
<th>Time* Fire treatment df=4,34</th>
<th>Time* Fire treatment * Habitat df=8,34</th>
</tr>
</thead>
<tbody>
<tr>
<td>Species richness</td>
<td>9.954***</td>
<td>14.791***</td>
<td>2.898***</td>
<td>14.033***</td>
<td>2.927*</td>
<td>2.610*</td>
</tr>
<tr>
<td>Bird abundance</td>
<td>22.991***</td>
<td>3.737*</td>
<td>2.457</td>
<td>1.867</td>
<td>7.336***</td>
<td>1.839</td>
</tr>
</tbody>
</table>
Table 3. BACI ANOVA F-values for the number of trees and shrubs per quadrat and the proportion of rubber vine in the understorey. No significant interaction terms were detected. Significant values are highlighted in bold (* $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$). Untransformed means (± 95%CI) for fire treatments and habitat are shown. Letters next to fire treatment means indicate differences or similarities of means based on post hoc Tukey HSD tests ($\alpha < 0.05$).

<table>
<thead>
<tr>
<th>Vegetation</th>
<th>$F$-values $\nu_{\nu-34}$</th>
<th>Fire treatment means</th>
<th>Habitat means</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$\nu_{\nu-2}$</td>
<td>$\nu_{\nu-1}$</td>
<td>Fire treatment</td>
</tr>
<tr>
<td>Trees / quadrat</td>
<td>0.437</td>
<td>2.292</td>
<td>6.577*</td>
</tr>
<tr>
<td>Shrubs / quadrant</td>
<td>0.188</td>
<td>6.839**</td>
<td>3.164</td>
</tr>
<tr>
<td>Rubber vine %</td>
<td>0.751</td>
<td>7.101**</td>
<td>146.944***</td>
</tr>
</tbody>
</table>
Table 4. Pearson correlations ($r$) of mean bird abundance, species richness, abundance of feeding groups and abundance of species that responded significantly to the second fire with the mean number of trees, shrubs, proportion of rubber vine and area of currant bush per quadrat. Significant values are highlighted in bold (* $P < 0.5$, ** $P < 0.01$, *** $P < 0.001$). Analyses used squared-root transformations for bird abundance, species richness, number of trees, shrubs and area of currant bush, log$_{10}(x + 1)$ transformation for abundance of feeding groups and individual species and arcsine transformation of the proportion of rubber vine.

<table>
<thead>
<tr>
<th></th>
<th>Before 2nd fire</th>
<th>1 year after 2nd fire</th>
<th>2 years after 2nd fire</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Tree</td>
<td>Shrub</td>
<td>Rubber</td>
</tr>
<tr>
<td>Abundance</td>
<td>-0.202</td>
<td>-0.402</td>
<td>-0.418</td>
</tr>
<tr>
<td>Species richness</td>
<td>-0.280</td>
<td>-0.380</td>
<td>-0.320</td>
</tr>
<tr>
<td>Carnivores</td>
<td>0.040</td>
<td>-0.466</td>
<td>-0.350</td>
</tr>
<tr>
<td>Frugivores</td>
<td>*<em>-0.587</em></td>
<td>0.351</td>
<td>-0.381</td>
</tr>
<tr>
<td>Great Bowerbird</td>
<td>0.105</td>
<td>-0.335</td>
<td>0.037</td>
</tr>
<tr>
<td>Mistletoebird</td>
<td>-0.382</td>
<td>0.080</td>
<td>-0.050</td>
</tr>
<tr>
<td>Granivores</td>
<td>0.110</td>
<td>0.015</td>
<td>-0.211</td>
</tr>
<tr>
<td>Red-winged Parrot</td>
<td>0.031</td>
<td>-0.123</td>
<td>-0.077</td>
</tr>
<tr>
<td>Insectivores</td>
<td>-0.279</td>
<td>-0.329</td>
<td>*<em>-0.494</em></td>
</tr>
<tr>
<td>White-throated HE</td>
<td>-0.078</td>
<td>-0.266</td>
<td>0.209</td>
</tr>
<tr>
<td>Nectarivores</td>
<td>0.041</td>
<td>-0.122</td>
<td>0.242</td>
</tr>
<tr>
<td>Noisy Friarbird</td>
<td>0.443*</td>
<td>0.001</td>
<td>-0.148</td>
</tr>
<tr>
<td>Rainbow Lorikeet</td>
<td>-0.168</td>
<td>-0.118</td>
<td>0.124</td>
</tr>
</tbody>
</table>

Short fire-return-intervals and birds
Fig. 1. Map of study region in northern Australia. First inset shows location of study sites along tributaries of the Burdekin River, north-eastern Queensland. Second inset (not to scale) shows fire treatment plots and habitat for Bend Creek (dashed line indicates separation of habitats; R = riparian, W = adjacent woodland). Fire treatment placement was randomly imposed along other creeks.
Fig. 2. Mean number of species (± 95%CI) per site observed in fire treatments over time in a) riparian habitat, and b) woodland habitat. Note that in a) the unburnt fire treatment in the ‘‘1 year after 2nd fire’ time period is obscured by the single burnt fire treatment.
Fig. 3. Mean bird abundance (± 95% CI) per site in fire treatments over time. Letters indicate differences between fire treatments within each year of survey based on post-hoc Tukey HSD tests (α < 0.05). Note that the single burnt fire treatment in the ‘Before 2nd fire’ time period is obscured by the twice burnt fire treatment.
Fig. 4. Canonical Discriminant Analysis of feeding group in fire treatments over time. 

a-c) The position of each site (small symbols) and centroids (large symbols) of fire treatments (woodland and riparian sites separated) displayed in each year. Dashed lines define the range of sites within each fire treatment*year combination. d) Centroids for each group. Arrows show change in centroid over time. e) The correlation between feeding groups and the canonical value.
Fig. 5. Untransformed mean feeding group abundance (± 95%CI) of a) frugivores, b) insectivores, c) nectarivores, d) granivores and e) carnivores per site in fire treatments over time. Letters within graph indicate significant differences between fire treatments within each year of survey based on post-hoc Tukey HSD tests (α < 0.05) and the asterisk (*) indicates that there was a strong trend (0.06 < P ≥ 0.05) for the twice burnt sites to have lower abundances than the unburnt sites.

Short fire-return-intervals and birds
Fig. 6. Associations of mean frugivore abundance per site and the mean amount of currant bush per quadrat within a site (10 x 20m) following the second fire. Lines on graph show linear trend lines (with the $r^2$ value displayed).
Appendix 1. ANOVA F-values for select species from 3-way ANOVA within each year of survey. Significant values are in bold (* $P < 0.5$, ** $P < 0.01$, *** $P < 0.001$) and values approaching significance are identified (^ $0.06 > P \geq 0.05$). Letters beside significant values indicate results from post-hoc Tukey HSD tests (fire treatment: U = unburnt, S = singularly burnt, T = twice burnt) or which habitat had highest abundances (habitat: W = woodland, R = riparian). Species with a significant response to the 2nd fire are highlighted in bold. F-values for the blocking factor Creek are not shown.

<table>
<thead>
<tr>
<th></th>
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<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Fire treatment $df = 2,10$</td>
<td>Habitat $df = 1,10$</td>
<td>Fire treatment $df = 2,10$</td>
</tr>
<tr>
<td>Carnivores</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pied butcherbird</td>
<td>11.976** S &amp; T &gt;U</td>
<td>124.116*** W</td>
<td>11.976**</td>
</tr>
<tr>
<td>Grey butcherbird</td>
<td>0.454</td>
<td>33.843*** R</td>
<td>2.885</td>
</tr>
<tr>
<td>Frugivores</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Great bowerbird</td>
<td>0.493</td>
<td>0.281</td>
<td>0.493</td>
</tr>
<tr>
<td>Mistletoebird</td>
<td>0.683</td>
<td>0.155</td>
<td>1.739</td>
</tr>
<tr>
<td>Granivores</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Red-winged parrot</td>
<td>0.679</td>
<td>0.642</td>
<td>1.348</td>
</tr>
<tr>
<td>Peaceful dove</td>
<td>2.156</td>
<td>0.111</td>
<td>1.089</td>
</tr>
<tr>
<td>Pale-headed rosella</td>
<td>0.825</td>
<td>0.770</td>
<td>0.377</td>
</tr>
<tr>
<td>Insectivores</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black-faced cuckoo-shrike</td>
<td>4.750* T &gt; U</td>
<td>0.101</td>
<td>4.273*</td>
</tr>
<tr>
<td>Dollarbird</td>
<td>1.980</td>
<td>3.337</td>
<td>0.064</td>
</tr>
<tr>
<td>White-throated gerygone</td>
<td>1.517</td>
<td>0.496</td>
<td>0.585</td>
</tr>
<tr>
<td>Magpie-lark</td>
<td>0.285</td>
<td>2.185</td>
<td>0.352</td>
</tr>
<tr>
<td>Australian magpie</td>
<td>0.118</td>
<td>8.857* W</td>
<td>0.048</td>
</tr>
</tbody>
</table>

Short fire-return-intervals and birds
**Red-backed fairy-wren** 3.888^* U > S & T 1.927 1.173 2.318 **5.163* W** 2.079 0.424 7.809^* W 0.424

**Yellow-throated miner** 1.728 **5.293* W** 0.092 2.270 3.845 2.270 1.221 0.010 0.561

**White-throated honeyeater** 0.152 **5.636* R** 2.843 **4.786* U > T** 0.283 1.707 **27.022*** S & U > T 2.157 0.749

**Olive-backed oriole** 0.333 1.748 0.199 1.731 1.199 0.127 1.315 0.322 0.902

**Striated pardalote** **4.402* S & T > U** 1.115 0.673 0.172 0.080 0.189 1.251 2.256 1.587

**Weebill** 0.898 **5.560* W** 0.428 0.121 **9.574* W** 0.092 3.162 **6.134* W** 0.278

**Apostlebird** 2.431 0.170 1.366 2.895 0.258 0.841 2.576 **4.896* R** 1.025

**Nectarivores**

**Blue-faced honeyeater** 1.404 0.047 0.145 0.104 0.569 3.221 0.758 3.737 0.351

**Yellow honeyeater** 2.054 3.552 0.450 2.284 1.247 0.538 3.328 3.280 1.590

**Little friarbird** 2.506 0.424 3.994^* 0.037 0.110 1.432 0.454 2.286 0.122

**Noisy friarbird** 2.422 0.286 0.074 **7.329* S & U > T** 1.538 0.582 0.776 0.551 0.053

**Rainbow lorikeet** 0.536 1.938 2.794 2.021 0.008 0.365 **4.559* S > T** 1.014 0.506

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Short fire-return-intervals and birds
Short fire-return-intervals and birds