

CHAPTER 3: SEASONAL INTERACTIONS BETWEEN WOODLAND BIRDS  
AND AREA OF REMNANT, DOMINANT TREE SPECIES, AND  
THE USE OF STRIPS OF VEGETATION AS FLY-WAYS

### 3.1 INTRODUCTION

Woodland clearing and fragmentation has altered the distribution and abundance of birds on the Armidale Plateau, an upland region of the New England Tablelands in northeastern New South Wales (Ford and Bell 1981; Howe 1984, 1986; Ford *et al.* 1985; Dunkerley 1989; Ford and McFarland 1991; Ford *et al.* 1994). Generally, open-country species have increased in abundance and woodland-dependent species, including those dependent on large areas of undisturbed woodland, have become less common (Howe 1986; see also Chapter 4). Numerous studies have emphasised the importance of habitat area as a predictor of bird species richness (Wilson and Willis 1975; Higgs and Usher 1980; Simberloff and Abele 1982), and to some extent this applies to bird assemblages on the Armidale Plateau (Howe 1984; see also Chapter 2). However, when a single survey of birds on transects of fixed area is considered, the area of the woodland patch is a negative predictor of species richness (Chapter 2). A closer examination indicates that the number of bird species per 1.2 hectare transect was greatest in intermediate-sized woodland patches, decreasing in patches that were larger than 400 ha or smaller than six hectares. This pattern was present during spring (1990) in woodland remnants on the Armidale Plateau. The present chapter investigates whether bird species richness is greatest in intermediate-sized woodland patches, during all seasons. This analysis also addresses the broader question of whether relationships between bird species richness and area, undergo seasonal changes.

While little is known about seasonal responses of birds to the size of remnants, the response of birds to seasonal habitat changes has been recorded in fragmented

landscapes in Australia (Cale 1990; Lynch and Saunders 1991; Woinarski and Tidemann 1991) and elsewhere (Morgan and Gates 1982; Arnold 1983; Winker *et al.* 1992). These studies suggest that in disturbed landscapes, birds make regular seasonal movements between patches of remnant vegetation. A similar situation appears to be occurring on the Armidale Plateau. The proportion of stringybark woodland or gum-box woodland was a significant predictor of abundance for a number of bird groups (Chapter 2). It appears that the relationship between bird groups such as the honeyeaters, and woodland type, is influenced by the alternate flowering of mistletoe species. In this chapter, the question of whether honeyeater species do make regular seasonal movements between remnants of gum-box and stringybark woodland on the Armidale Plateau is investigated. In these two woodland types, bird assemblages and the seasonal variation in bird abundance are compared.

It is widely proposed that most seasonal and daily bird movements should continue in a fragmented landscape, provided the remnants are linked by strips of vegetation which can serve as corridors (Bennett 1991, 1992; Saunders and de Rebeira 1991; Saunders and Hobbs 1991; Merriam and Saunders 1993). It is also possible that birds use riparian vegetation as fly-ways (Finch 1985; Croonquist and Brooks 1993; Isaacs 1994). Yet data on the movement of wildlife along corridors are scarce. Little is known about the extent to which birds move through a disturbed landscape, and whether strips of vegetation are important (Simberloff *et al.* 1992). In the Western Australian wheatbelt, patches of remnant vegetation have discrete boundaries and the area between remnants is virtually treeless. Such a landscape is truly fragmented, and it appears that birds must stay within patch boundaries, or within strips of vegetation when moving between patches of remnant vegetation (Saunders and Ingram 1987; Lynch *et al.* in press). By contrast, the Armidale Plateau is a variegated rather than a fragmented landscape (McIntyre and Barrett 1992; see also Chapters 4 and 5). The woodland remnants are diffusely fragmented and often hard to delineate. In such a landscape, strips of woodland may be relatively unimportant for birds moving between remnant patches,

since birds have the option of using the scattered trees between the remnants as 'stepping stones'. It is also questionable whether strips of vegetation will act as thoroughfares for bird movement in landscapes where there is an increased abundance of edge species, including predators (Simberloff *et al.* 1992).

In particular, Catterall *et al.* (1991) have questioned the value of strips of vegetation as corridors, in environments where aggressive territorial species like noisy miners *Manorina melanocephala* are common. This species is common on the Armidale Plateau as in other landscapes that have been over-cleared and over-grazed (Loyn 1987; Grover and Slater 1994; Grey 1995). As well as possibly preventing other bird species from using strips of vegetation as thoroughfares for movement, the increased abundance of noisy miners has been implicated in the widespread decline of bird species richness on the Armidale Plateau (Ford 1985; see also Chapter 2). The effect of noisy miners on bird assemblages on the Armidale Plateau is examined further in this chapter, and the abundance of individual bird species is compared inside and outside of miner territories. The temporary establishment of two noisy miner territories and resulting decline in bird species richness is also described. Bird species richness is also compared each season in patches of woodland, strips of woodland, grassland, and along water-courses, and the extent to which birds use these different sites as fly-ways is assessed.

### 3.2 STUDY AREA

The New England Tablelands were first settled by graziers in the 1830s. Although clearing of woodland continues, extensive clearing had stopped by the 1930s (Davidson and Davidson 1992). About half of the area on the Tablelands is cleared, while the remainder is covered by dry sclerophyll woodland or open forest (Smith and Turvey 1977). The study site, which will be referred to as the Armidale plateau, is the upland area within a 60 km radius of Armidale (151° 40' E, 30° 32' N). It is intensively grazed and has been more extensively cleared than the rest of the New England Tablelands,

with the remaining tree cover being less than 30%. Most remaining tree cover on the Armidale Plateau is restricted to rocky, steep country on poor soils. About half of this woodland is affected by eucalypt dieback, which results in widespread tree death (Jones *et al.* 1990). The two most common types of woodland are New England stringybark *Eucalyptus caliginosa*, occurring on the poorer rocky soils along ridges and hill-tops, and the Blakely's red gum *E. blakelyi* and yellow box *E. melliodora* association, restricted to the richer soils at lower altitudes. It is the gum-box woodland that has been most intensively cleared. The understorey is sparse throughout the study region, either naturally or due to grazing and clearing. The common understorey species are the fern-leaf wattle *Acacia filicifolia*, black thorn *Bursaria spinosa* and sifting bush *Cassinia quinquefaria*, as well as regenerating eucalypts. Elevation of the study area ranges from 900 m to about 1300 m. The climate is temperate, with an average annual rainfall of 795 mm (a monthly range of 45 to 100 mm) and a yearly temperature range from 6 °C in July to 21°C in January (average monthly temperatures).

### **3.3 METHODS**

#### **3.3.1 Bird Survey**

Birds were surveyed using strip transects 60 m wide and of varying length. The rate of movement along the transect was constant at 5 minutes per 50 m. This method is the same as described in Chapter 2 (see also Barrett *et al.* 1994). Birds were surveyed each season for one or two years (4 to 8 surveys). In this study a single survey site may be used in more than one comparison. For example, sites used to compare the effects of woodland patch size may also have been used to compare the effects of water-courses, noisy miner territories and dominant tree species on bird assemblages.

### 3.3.2 Area of Woodland Patch

The number of bird species was counted each season for two years, in 50 strip transects. The 60 x 200 m transects were randomly placed throughout woodland patches ranging in size from a few hectares to greater than 400 ha. Roughly two thirds of the 294 survey sites occurred with at least one other survey site in the same woodland patch. Up to 14 different survey sites occurred in patches which were larger than 400 ha in area, however, the average number in a single, continuous woodland area was between three and four sites. The distance to the nearest site in the same patch varied from 100 metres to about 2000 metres with the overall average being between 250 and 300 metres.

Patch area (ha) was calculated using both 1:25 000 topographic maps and aerial photos. The majority of sites selected were in stringybark woodland as this woodland is most common throughout the study area. For the analysis, patches were grouped into five size categories; greater than 400 ha (n = 12 transects), 101-400 ha (n = 12 transects), 21-100 ha (n = 15 transects), 6-20 ha (n = 5 transects) and less than 6 ha (n = 6 transects). Sites in which noisy miners were present during at least five of the eight surveys were excluded from the analysis of area effects (see the analysis section below for an explanation of the statistical techniques used).

### 3.3.3 Dominant Tree Species

The number of bird species and the total number of honeyeaters (11 species) were compared in stringybark woodland and gum-box woodland in each season for two years. Six strip transects varying in length from 200 m to 350 m, were placed in similarly sized patches of both woodland types (the total length of the transects was 1500 m for each woodland type). The average proportion of stringybarks ( $\pm$ SE) on transects in stringybark woodland was  $0.95 \pm 0.01$ , and the average proportion of gum and box trees in gum-box woodland was  $0.94 \pm 0.02$ . The average size ( $\pm$  SE) of the patches was  $31 \text{ ha} \pm 1.2$  for the stringybark woodland and  $30 \text{ ha} \pm 1.4$  for the gum-box woodland. Friedman's non-parametric, two-way analysis of variance (Sokal and Rohlf 1981) was

used to test for differences in the abundance of individual species in the different woodland types. In this analysis, for each of the 8 surveys the mean number of birds per hectare (for the six transects in each treatment) was calculated for each bird species. These eight means were then treated as repeated measures (see Wilkinson *et al.* 1992 for an explanation of these procedures).

### **3.3.4 Woodland Patches, Water-courses, Corridors, Peninsulas and Grassland**

The number of bird species as well as the total number of birds, was estimated each season for one year on transects in woodland patches, along water-courses, in strips of woodland and in grassland (Figure 3.1). Strips of woodland were defined as being less than 100 m wide and at least 300 m long. Strips that connected two patches of woodland greater than 40 ha in area were defined as corridors, and strips that projected into the grassland from a single woodland patch (greater than 40 ha in size) were defined as peninsulas. Transects were placed in three corridors ( $n = 3$  transects, average length = 433 m) and three peninsulas ( $n = 3$  transects, average length = 950 m). Two of the three peninsulas were along scaled roads, the remaining peninsula and the three corridors were on farmland. Seven transects were also placed in seven adjoining or nearby woodland patches (average length = 457 m), and four transects were placed in the grassland immediately adjacent to the strip or patch (average length = 450 m). For comparison, six transects were also placed along six water-courses within woodland patches (average length = 450 m).

### **3.3.5 Fly-ways**

The use of strips of woodland as fly-ways was investigated by recording the number of long-distance flights ( $> 40$  m in length, all bird species combined) along and across transects in corridors, peninsulas and patches of woodland. Flights along and across the transect were also recorded on transects adjacent to water-courses and on transects in grassland. Water-course transects were oriented along the water-courses, as were transects along corridors and peninsulas. Transects in patches and grassland were

generally perpendicular to the edge of the woodland patch (Figure 3.1). It was expected that if birds were using strips of vegetation as fly-ways, the total number of flights (> 40 m) would be greater on the transects in corridors and peninsulas, than on transects in patches or grassland. Furthermore, flights would tend to be along rather than across the transect in strips of woodland, whereas on transects in patches of woodland or in grassland no difference would be expected between the number of flights along and across the transect. It was also expected that if birds were actively seeking out corridors and peninsulas when they were moving through the landscape, the proportion of flights to the number of birds recorded would be greater on transects in corridors and peninsulas. The number of flights (> 40 m) per hectare was compared between seasons for one year. For individual bird species, seasonal surveys from two years (8 surveys) were used to estimate the relative number of flights along transects in corridors and peninsulas. In this analysis no distinction is made between flights by individual birds that were flying alone or as part of a flock. Flights that were more than 40 m above the upper canopy were not recorded. Transects along water-courses were all within patches of woodland. This allows the use of water-courses as fly-ways to be examined independently of woodland cover.

### **3.3.6 Noisy Miners**

Birds were surveyed for 20 minutes in each season over a two year period (8 surveys) in 63 strip transects. All transects were 200m long and 60 metres wide, and placed randomly within patches of woodland, that varied in size from a few hectares to greater than 400 ha. Transects were identified as being inside noisy miner territories if miners were recorded during at least five of the eight surveys. The total number of bird species in miner sites was compared with non-miner sites. Seasonal variation in the number of noisy miners was also analysed.

During the first winter of the two year study (winter 1991), two noisy miner territories were temporarily established on separate transects, within a single 40 ha woodland

patch. One territory was temporarily established on a 100 m section of a transect (the number of noisy miners increased from zero to 18.3 miners / ha) and a second miner territory was established on 150 m of a second transect (the number of miners increasing from zero to 16.7 miners / ha). In order to estimate the impact that the establishment of a noisy miner territory had on bird species richness, similarly sized transects that continued to be unoccupied by miners were selected as controls. In both the control and 'experimental' transects, two surveys had been carried out prior to (spring 1990 / summer 1991), and after (spring 1991 / summer 1992) the miner territories being established. No censuses were carried out in any sites during autumn 1991. During the winter period (1991) when the miner territories were present, two surveys were carried out, a month apart. The cumulative number of bird species from both surveys on the two replicate transects was calculated before, during and after the presence of miners in each of the control and experimental treatments.

### 3.3.7 Analysis

The effects of a particular environmental variable, as well as seasonal and yearly effects, were analysed using repeated measures analysis of variance (Tabachnick and Fidell 1989; Wilkinson *et al.* 1992). Where transects were of unequal length, it was necessary to convert counts to number per hectare before the analysis. In such cases the length of the transect (m) was included as a covariate after testing for homogeneity of slopes, and then the analysis was carried out using the adjusted mean squares (Tabachnick and Fidell 1989; Wilkinson *et al.* 1992). As a result, there were small differences between the results of the analyses which were done using the adjusted cell means and the untransformed counts per hectare presented in the figures. In all cases where transects were of unequal length, the total transect length was the same across all treatments. All data were log-transformed to improve normality and homogeneity of variances. In some instances the Greenhouse-Geiser correction statistic (G - G) was used as a precaution against making type 1 errors (LaTour and Miniard 1983; Wilkinson *et al.* 1992). *A posteriori* tests for differences between means were carried out using Student Newman-



Kuels test (SNK) for repeated measures (Winer 1971). Chi-square, Friedman's and Mann-Whitney U-tests (Sokal and Rohlf 1981), and multivariate repeated measures analysis of variance (MANOVA) were also used (Wilkinson *et al.* 1992).

### 3.4 RESULTS

#### 3.4.1 Area of Patch

There was no overall difference in the number of bird species on transects in the different-sized woodland patches or between years (Figure 3.2, years combined, patch size;  $F = 1.4$ , d.f. = 4, 45,  $p = 0.25$ , year;  $F = 0.02$ , d.f. = 1, 45,  $p = 0.9$ ). However, there was significant seasonal variation ( $F = 10.7$ , d.f. = 3, 135,  $p < 0.01$ ) and a significant interaction between woodland patch size and season (Wilks' Lambda = 0.56,  $F = 2.0$ , d.f. = 12, 114,  $p = 0.01$ ). *Post hoc* tests indicate that there were significantly more bird species present during spring than winter in patches between 100 and 400 ha (Figure 3.2, compare a and b, SNK for adjusted means; year 1;  $q = 6.5$ ,  $r = 33$ , year 2;  $q = 6.06$ ,  $r = 35$ , d.f. = 135,  $p < 0.05$ ). There were no other significant differences, although the increased species richness in patches between 100 and 400 ha was also evident during the autumn months (Figure 3.2).

When spring and autumn were combined and analysed separately, the area of the woodland patch had a significant effect on the number of bird species (years combined,  $F = 3.0$ , d.f. = 4, 45,  $p < 0.05$ ). Although the SNK tests show no differences ( $p > 0.05$ ), the number of species was greater in patches that were between 6 and 400 ha in size during both spring and autumn and lower in the largest and smallest patches (Figure 3.2). When summer and winter were combined and analysed separately, the area of the woodland patch had no significant effect on the number of bird species (years combined,  $F = 1.7$ , d.f. = 4, 45,  $p > 0.05$ ).

### 3.4.2 Dominant tree species

There was no significant difference in the total number of bird species per hectare in stringybark woodland when compared with equal sized patches of gum-box woodland. This was true whether miner sites were included ( $n = 6$  transects;  $F = 1.94$ , d.f. = 1, 10,  $p = 0.19$ ) or excluded from the analysis ( $n = 2$  transects;  $F = 7.52$ , d.f. = 1, 2,  $p = 0.11$ , four of the six transects in gum-box woodland were in noisy miner territories). There was also no difference in the total number of birds per hectare in stringybark woodland when compared with gum-box woodland (miner sites included,  $n = 6$ ,  $F = 0.01$ , d.f. = 1, 10,  $p = 0.96$ ). Noisy miners were more common in the gum-box woodland (Friedman's  $X^2 = 4.5$ , d.f. = 7,  $p < 0.05$ , Figure 3.3c), as were Australian magpies *Gymnorhina tibicen* (Friedman's  $X^2 = 8.0$ , d.f. = 7,  $p < 0.05$ ). The following seven species were significantly more common in stringybark woodland (Friedman's Test,  $p < 0.05$ , d.f. = 7); striated thornbill *Acanthiza lineata* ( $X^2 = 8.0$ ), scarlet robin *Petroica multicolor* ( $X^2 = 4.5$ ), grey fantail *Rhipidura fuliginosa* ( $X^2 = 4.5$ ), buff-rumped thornbill *Acanthiza reguloides* ( $X^2 = 4.5$ ), orange-winged sittella *Daphoenositta chrysoptera* ( $X^2 = 4.5$ ), spotted pardalote *Pardalotus punctatus* ( $X^2 = 4.5$ ) and the white-naped honeyeater *Melithreptus lunatus* ( $X^2 = 6.13$ , Figure 3.3a).

The number of honeyeater species showed a significant interaction between woodland type and season (Figure 3.4a and b,  $n = 6$ ,  $F = 4.5$ , d.f. = 3, 30,  $G-G < 0.05$ ). *Post hoc* multiple comparison tests within each season showed no significant differences in the number of honeyeater species in stringybark woodland when compared with gum-box woodland ( $p > 0.05$ ). However, honeyeater species were more common on transects in gum-box woodland than stringybark woodland during summer, and showed the reverse pattern during winter. This trend may be the result of migratory species, such as the white-naped honeyeater and yellow-faced honeyeater *Lichenostomus chrysops*, moving into stringybark woodland during the winter months and being largely absent from the Tablelands in summer (Figure 3.3c and b). The pattern may also be due to resident species such as the fuscous honeyeater *Lichenostomus fuscus*, being more common in

the gum-box woodland during spring and summer, but appearing to shift into the stringybark woodland during winter (Figure 3.3d, differences not significant). Other less common honeyeater species recorded on the transects were; the red wattlebird *Anthochaera carunculata*, noisy friarbird *Philemon corniculatus*, white-plumed honeyeater *Lichenostomus pencillatus*, brown-headed honeyeater *Melithreptus brevirostris*, tawny-crowned honeyeater *Phylidonyris melanops*, eastern spinebill *Acanthorhynchus tenuirostris* and the Silvereeye *Zosterops lateralis* (for the purposes of this study, the silvereeye was defined as a honeyeater).

### 3.4.3 Woodland Patches, Strips, Water-courses and Grassland

There was a significant difference in the number of bird species between transects in woodland patches, along water-courses, corridors, peninsulas and in grassland (Figure 3.5a,  $F = 16.4$ , d.f. = 4, 17,  $p < 0.01$ ). During summer, autumn and winter the number of bird species was significantly greater on transects in patches than those in grassland (Figure 3.5a, SNK for adjusted means; summer;  $q = 5.13$ ,  $r = 11$ , d.f. = 51; autumn;  $q = 5.79$ ,  $r = 14$ , d.f. = 51 and winter;  $q = 5.05$ ,  $r = 12$ , d.f. = 51, all  $p < 0.05$ ). In summer the number of species was also greater along water-courses than in grassland (Compare \* in Figure 3.5a; SNK for adjusted means;  $q = 5.84$ ,  $r = 14$ , d.f. = 51,  $p < 0.05$ ). There was no significant seasonal variation in the number of bird species within any of the types of sites. In patches of woodland, there was an average of 3.8 species per hectare (all seasons combined). Along water-courses the average was 3.7 species per hectare, in corridors 2.8 species per hectare, in peninsulas 1.9 species per hectare and in grassland 1.1 species per hectare. Species occurring in strips of woodland were a subset of those occurring in patches.

There was a significant difference in the total number of birds per hectare (all species combined) between transects in patches, along water-courses, corridors, peninsulas and in grassland (Figure 3.5b,  $F = 13.4$ , d.f. = 4, 17,  $p < 0.01$ ). The number of birds was significantly greater on corridor transects than on grassland transects during spring and

winter (Figure 3.5b, SNK for adjusted means; spring corridor vs grassland;  $q = 6.04$ ,  $r = 18$ ,  $d.f. = 51$ , winter corridor vs grassland;  $q = 5.4$ ,  $r = 12$ ,  $d.f. = 51$ , both  $p < 0.05$ ). The variation in the number of birds was also greater along corridors during spring. This was partly due to birds flocking. The number of birds was also greater along water-courses during summer than on grassland transects during summer (Compare \* in Figure 3.5b, SNK for adjusted means;  $q = 6.7$ ,  $r = 16$ ,  $d.f. = 51$ ,  $p < 0.05$ ). There was no significant difference between sites in autumn, although the number of birds per hectare was greatest in corridors. There was no significant seasonal variation in the number of birds within any of the sites. As with species richness, seasonal differences were most obvious on transects in corridors. In contrast to the trend shown by bird species richness (Figure 3.5a), the number of individuals per hectare was as high in strips of woodland (corridors and peninsulas) as it was in patches of woodland.

#### 3.4.4 Fly-ways

There was a significant difference in the total number of flights per hectare (greater than 40 m in length) between transects in patches, along water-courses, corridors, peninsulas and in grassland (Figure 3.6,  $F = 4.7$ ,  $d.f. = 4, 17$ ,  $p = 0.01$ , seasons combined). *Post hoc* SNK tests indicate that the total number of flights was significantly greater in corridors than in grassland ( $q = 5.9$ ,  $r = 5$ ,  $d.f. = 17$ ,  $p < 0.05$ ). There were no significant seasonal effects on the number of flights along or across the transects (Figure 3.7a to d,  $F = 1.9$ ,  $d.f. = 3, 105$ ,  $p > 0.05$ ). Overall, there were more flights along than across transects ( $F = 12.7$ ,  $d.f. = 1, 35$ ,  $p < 0.01$ ). *Post hoc* SNK tests indicate that there were no significant differences in the number of flights across transects within each season, regardless of whether the transect was in a corridor, peninsula, patch or on grassland (Figure 3.7a to d).

The number of flights along transects was significantly greater in corridors than in patches or grassland during summer (Figure 3.7b, SNK for adjusted means; corridor vs patch;  $q = 5.6$ ,  $r = 30$ ,  $d.f. = 150$ , corridor vs grassland;  $q = 5.45$ ,  $r = 33$ ,  $d.f. = 150$ , both

$p < 0.05$ ). During autumn the number of flights along transects in corridors was significantly greater than the number along transects in patches, along water-courses and along transects in grassland (Figure 3.7c, SNK for adjusted means; corridor vs patch;  $q = 5.8$ ,  $r = 9$ , d.f. = 150, corridor vs water-course;  $q = 5.8$ ,  $r = 10$ , d.f. = 150, corridor vs grassland;  $q = 6.8$ ,  $r = 26$ , d.f. = 150), all  $p < 0.05$ ). Although *post hoc* tests showed no significant differences, the proportion of flights along transects was greater along water-courses, corridors and peninsulas during spring (Figure 3.7a), and also greater in corridors during winter (Figure 3.7c). These results indicate that flights are more likely to be along transects when the transects occur in strips of woodland, than if they are in patches of woodland or grassland. However, the proportion of flights to number of birds showed no significant differences between woodland patches, water-courses, corridor, peninsulas or grassland ( $F = 1.3$ , d.f. = 4, 17,  $p > 0.05$ , flights along and across combined). This indicates that the birds are not actively seeking strips of vegetation at times when they are more likely to be flying long distances.

There was a significant interaction between the direction of the flight (along or across the transect) and the site of the transect (Figure 3.7a to d,  $F = 8.2$ , d.f. = 4, 35,  $p < 0.01$ ). *Post hoc* tests only indicate that there were significantly more flights along than across transects in corridors during autumn (Figure 3.7c, SNK for adjusted means;  $q = 6.1$ ,  $r = 22$ , d.f. = 150,  $p < 0.05$ ). Generally, flights were more likely to be along transects in strips of vegetation (corridors and peninsulas). There was also an interaction between the direction and the season of the flight (Figure 3.7a to d,  $F = 3.5$ , d.f. = 3, 105,  $G-G < 0.05$ ). *Post hoc* tests found no significant differences, however, this interaction was most obvious along water-courses. A greater proportion of the flights were along transects during spring, and to a lesser extent autumn when birds are more likely to be moving through the landscape (Figure 3.7a and c). During summer and winter the proportion of flights along and across the transects were similar (Figure 3.7 b and d). These seasonal shifts in direction of flight suggest that water-courses are used as flyways during spring and to a lesser extent autumn (Figures 3.7a and c). The interaction

between the direction of flight and season was less obvious in strips of vegetation because, being surrounded by grassland, birds were less likely to fly across the transects in strips of woodland. There was no significant three-way interaction between habitat, season and direction (Figure 3.7a to d,  $F = 0.5$ , d.f. = 12, 105,  $G-G > 0.05$ ).

A total of 574 flights by 35 species were recorded along the six transects in strips of woodland (corridors and peninsulas) over two years (8 seasonal surveys). Eastern rosellas *Platycercus eximius* accounted for 32% of these flights, noisy miners - 18%, brown-headed honeyeaters 9% and Australian magpies - 5%. If noisy miners and open-country species such as the eastern rosella, red-rumped parrot *Psephotus haematonotus*, Australian magpie and Australian raven *Corvus coronoides* are excluded, there were 216 flights along these same transects. Brown-headed honeyeaters accounted for 23% of these 216 flights, the crimson rosella *Platycercus elegans* 11%, red wattlebird 9%, striated thornbill 8%, the black-faced cuckoo-shrike\* *Coracina novaehollandiae* and noisy friarbird\* 6% respectively, pied currawong\* *Strepera graculina* 5%, striated pardalote *Pardalotus striatus* 4%, yellow-faced honeyeater\*, white-naped honeyeater\* and wood duck *Chenonetta jubata* all 3% (\* migrant species - that is, species which show definite seasonal changes in abundance on the Armidale Plateau. Note that non-migrant species also move 'opportunistically' throughout the landscape). Of the 35 species recorded flying along strips of woodland, 15 (43%) were either summer, winter or partial migrants and the remainder were resident species. This compares to 9 migrant species out of 24 species (37%) recorded flying along transects in patches of woodland, and 5 migrant species out of 25 species (20%) recorded flying along water-courses.

### 3.4.5 Noisy Miners

Of the 63 transects surveyed, 13 were identified as being inside noisy miner territories. The total number of bird species was significantly lower on these 13 transects compared with the remaining 50 sites (Mann-Whitney;  $U = 490.5$ ,  $p < 0.01$ ). The following seven species were significantly less common (Mann-Whitney,  $p \leq 0.05$ ) in noisy miner sites;

striated thornbill ( $U = 439.0$ ), white-throated treecreeper *Climacteris leucophaea* ( $U = 470.5$ ), white-naped honeyeater ( $U = 478.5$ ), spotted pardalote ( $U = 551.5$ ), grey shrike-thrush *Colluricincla harmonica* ( $U = 476.0$ ), yellow-faced honeyeater ( $U = 446.5$ ) and the eastern spinebill ( $U = 455.0$ ). In addition to noisy miners ( $U = 131.0$ ), two open-country species; the eastern rosella ( $U = 131.0$ ) and galah *Cacatua roseicapilla* ( $U = 235.0$ ) were significantly more common in miner territories. Although not significant, grey butcherbirds were twice as common in noisy miner territories (inside mean = 1.2 birds / transect, outside mean 0.6 birds / transect, Mann-Whitney;  $U = 238.5$ ,  $p = 0.08$ ). The number of noisy miners on the 13 transects in miner sites was greatest during the spring and lowest during summer ( $F = 5.6$ , d.f. = 3, 36, G-G < 0.01, SNK for adjusted means; d.f. = 36,  $q = 4.1$ ,  $p < 0.05$ ). The total number of bird species in noisy miner sites was also greatest during spring, and lowest during winter ( $F = 5.6$ , d.f. = 3, 36, G-G < 0.01, SNK for adjusted means; d.f. = 36,  $q = 4.4$ ,  $p < 0.05$ ). There was no significant interaction between the total number of bird species in noisy miner sites and season ( $F = 0.3$ , d.f. = 3, 183,  $p = 0.9$ ).

#### 3.4.6 Temporal Changes in Noisy Miner Territories

The average number of bird species per hectare (2 transects, 2 censuses;  $n = 4$ ) decreased on the experimental transects after the arrival of noisy miners (Figure 3.8). The cumulative number of bird species on the two transects where noisy miners temporarily established a territory, decreased from 24 to 6 species and increased again to 21 after the miners had left (counts combined for both surveys in both transects;  $X^2 = 10.9$ , d.f. = 2, expected = 17.0,  $p < 0.01$ ). Over the same time period the cumulative species count for the two control transects decreased from 21 to 16 species and then increased to 21, but this difference was not significant ( $X^2 = 0.9$ , d.f. = 2, expected = 19.3,  $p > 0.05$ ). This small decrease in the number of bird species on control transects during the experimental period (see also Figure 3.8) reflected a general decline in bird species richness during winter. However, it may also be partly due to the close proximity of the control transects to the experimental transects (both control transects

were less than 200 m from the experimental transects). It is possible that the effect of the miner territories was detectable throughout the whole of the 40 ha woodland patch. If the cumulative species counts for both the experimental and control transects were combined into a two-way contingency table there was no significant difference between control and experimental treatments ( $X^2 = 4.5$ , d.f. = 2,  $p > 0.05$ ).

### 3.5 DISCUSSION

This chapter demonstrates that the pattern of increased bird species richness in intermediate-sized woodland patches described in Chapters 2 and 4, is to some degree seasonal. Bird species appear to move into, or through patches during spring and autumn. Conversely, birds appear to move out of these same patches, or avoid these patches during the summer and winter months (Figure 3.2). Spring and autumn are the migration periods for a number of species on the Tablelands (see Chapter 2 for migratory species). So it appears that during this time, when birds are more likely to be moving through the Armidale Plateau, intermediate-sized patches are favoured (although when two seasons were combined the difference was not significant). It may be that patches within this size range (particularly 100 to 400 ha) are most suitable as 'stop-over' remnants for migrating birds. Migrating or transient birds are likely to be more dispersed in the extensively wooded areas, and concentrated into remnants as a result of habitat fragmentation. This could explain the decreased species richness in remnants larger than 400 ha. These same species may be driven from, or may avoid the smallest remnants (< 6 ha), because of the increased presence of predators or territorial species, such as noisy miners in these sites (Chapter 2). Spring and autumn are also the periods when the young birds of resident species are most likely to be dispersing in search of new territories, and spring is the period when birds search for breeding habitat. Nest predation by birds tended to be greater in patches smaller than 100 ha, and nest predation by both birds and mammals tended to be greater in patches larger than 400 ha



(Barrett in prep.). As a result, the increased bird species richness in the intermediate-sized remnants during both spring, and to a lesser extent autumn, may be due in part to nest predator avoidance.

In a previous study on the New England Tablelands, Howe (1984) found that bird species assemblages in small habitat islands (0.1 to 7 ha) were generally more predictable over space and time, than species assemblages in large forest areas (> 500 km<sup>2</sup>). Howe (1984) concluded that this result was consistent with the theory of equilibrium biogeography (MacArthur and Wilson 1963) which predicts that rates of "immigration" and "emigration" will be greater in extensive forest areas than in small forest islands. Howe's (1984) study site was east of Walcha, south of the Armidale Plateau, where remnants tend to have discrete boundaries, and the grassland between the remnants is largely treeless. The island biogeography model may be appropriate for such a fragmented landscape. However, the present study suggests that it has limited application to the landscape on the Armidale Plateau which is variegated rather than fragmented (McIntyre and Barrett 1992; see also Chapters 2, 4 and 5). Rather than being greatest in the extensively wooded areas, the variation in species richness on the Armidale Plateau was greatest in the intermediate-sized remnants (particularly between 100 and 400 ha). It is possible that if Howe (1984) had surveyed birds in forest 'islands' that were larger than seven hectares, he may have found species turnover to have been greater in these larger 'islands', than in the control plots in the extensive forests areas (> 500 km<sup>2</sup>).

The predictive models previously presented (Chapter 2) suggest that bird species richness was positively associated with stringybark woodland, rather than gum-box woodland. Woodland type was not, however, a predictor of bird species richness when the models were constructed using bird survey data from repeated surveys (Chapter 2, Table 2.6). The analysis presented here is partly based on the same survey data, so it is not surprising that this latter result is confirmed. Although species richness did not

differ between woodland types, the noisy miner and Australian magpie were more common in gum-box woodland. A greater number of species were more common in stringybark woodland, which is consistent with the findings in Chapter 2. Ford *et al.* (1986) observed that striated thornbills, scarlet robins, grey fantails, buff-rumped thornbills, spotted pardalotes and white-naped honeyeaters spend a greater proportion of foraging time on stringybark trees rather than gum or box trees. This is consistent with the present study, which found these species to be more common in stringybark woodland. However, their study indicates that the number of species which show a preference for either gum-box woodland or stringybark woodland, was underestimated by the present study.

It is surprising that bird species richness was not greater in the gum-box woodland which occurs on richer soils at lower altitude (Recher 1985; Braithwaite *et al.* 1989; Recher and Majer 1994). It was suggested that this may be a result of the increased abundance of territorial species (Chapter 2). The present study confirms that noisy miners are more common in gum-box woodland. However, the present study also found that even if sites with miners were excluded from the analysis, species richness was still no greater in gum-box woodland than stringybark woodland. This is likely to be a result of the small sample size ( $n = 2$ ). However, it does suggest that other factors may also be involved. As was suggested in Chapter 2, it may be historical in that, the loss of gum-box woodland has been so great that there is no longer a complete avifauna. That is, many bird species associated with this woodland type have gone locally extinct. It is also likely that the removal of timber, particularly logs, from gum-box woodland, has made the stringybark woodland preferable to many species (Chapter 2).

The drooping mistletoe *Amyema pendulum* occurs in the stringybark woodland and flowers mostly during winter and spring. The box mistletoe *A. miquelii*, associated with the gum-box woodland, flowers mostly during summer and autumn (Reid unpublished data). Given that mistletoes are an important source of nectar for honeyeaters on the

Armidale Plateau (Ford *et al.* 1986), the alternate flowering of mistletoe species may partly explain the interaction between woodland type and season shown by the number of honeyeater species (Figure 3.4). This interaction is likely to be due in part to winter migrants, such as the white-naped and yellow-faced honeyeaters, visiting stringybark woodland to feed on nectar from the drooping mistletoe (Figure 3.3a and b). The interaction may also be due to a portion of the population of resident species, such as fuscous honeyeaters and to a lesser extent noisy miners, moving from the gum-box woodland in summer to the stringybark woodland in winter, in response to the alternate flowering of mistletoes (Figures 3.3c and d). This latter suggestion is supported by the temporary establishment of two noisy miner territories described in this study. Both territories were established over winter in stringybark woodland, where flowering mistletoes were numerous (unpublished data). Ford *et al.* (1990) described a seasonal correlation between the availability of flowering mistletoes, and the abundance of fuscous honeyeaters *Lichenostemos fuscus* in Eastwood State Forest, a 200 ha woodland patch on the Armidale Plateau. Seasonal movements by honeyeaters in response to flowering trees and shrubs, have been described in the Western Australian wheatbelt by Cale (1990) and Lynch and Saunders (1991). In the study by Cale (1990), the density of singing honeyeaters and brown honeyeaters was positively correlated with the quantity of flowers in shrubland-heath. Further, there appeared to be a regular movement of singing honeyeaters between shrubland-heath and mallee/shrubland, in response to the availability of flowers. The present study indicates that similar movements between habitats may be occurring on the Armidale Plateau in response to mistletoe flowering.

The greater variation in species richness in the intermediate-sized patches, suggests that these patches provide seasonal habitat for birds. This, as well as the interaction between season and woodland type shown by honeyeaters, indicate that a number of bird species are making regular seasonal movements among a large portion of the woodland remnants on the Armidale Plateau. Other authors have found that at certain times of the year, relatively small remnants including roadside vegetation, serve as habitat for

migratory or locally wandering bird species [Leach and Hines (1993) in southeast Queensland; Date *et al.* (1991) in northeast New South Wales; Cale (1990); Arnold and Weeldenburg (1990) and Lynch and Saunders (1991) in roadside vegetation in the Western Australian wheatbelt]. In the present study it was assumed that a bird seen flying in a straight line for a distance of at least 40 metres was more likely to be moving through the landscape (not necessarily a migratory species). If this assumption is valid, strips of woodland linking two woodland patches appear to act as thoroughfares for bird movement on the Armidale Plateau (Figure 3.6). If strips of woodland linking woodland patches are important to birds moving through a variegated landscape such as that on the Armidale Plateau, it is reasonable to suggest that corridors will be even more important for facilitating movement in a fragmented landscape, such as the Western Australian wheatbelt, where there are no scattered trees between remnants. The limited studies that are available suggest that this is so (Saunders and Ingram 1987; Lynch *et al.* in press).

The proportion of birds observed flying was similar in all sites, indicating that when birds are moving through the landscape they are not concentrated in corridors. However, the orientation of the flight did change depending on the site. Whereas flights in woodland patches and grassland were along and across the transect in equal portions, flights in strips of woodland were more likely to be along the transect (Figures 3.7a to d). This suggests that birds flying through the landscape stay within tree cover whenever possible. As a result, whether by conscious decision or by default, strips of vegetation act as thoroughfares for bird species. It is likely that both migrant and resident species stay within woodland cover when they are moving through the landscape, because they forage along the way. If this is so, then movement along strips of woodland may take place as a series of smaller flights (less than 40 m) from canopy to canopy. As a result, counting the number of flights greater in length than 40 metres, is likely to have underestimated the number of birds, particularly the number of smaller species, moving through wooded areas. Another likely explanation for birds avoiding

grassland, is that avian predators, such as the brown falcon *Falco berigora* and brown goshawk *Accipiter fasciatus* are common throughout the Armidale Plateau. There was a greater proportion of flights along, rather than across transects near water-courses during spring (Figure 3.7a). This suggests that even when woodland cover is continuous, linear habitat features such as water-courses within woodland patches are used by birds as flyways. This suggestion is supported by winter migrants being positively associated with water-courses on the Armidale Plateau (Chapter 2).

Catterall *et al.* (1991) found an increase in the number of aggressive bird species such as the noisy miner, in strips of urban forest in southeast Queensland. This appears to also be true for strips of vegetation on the Armidale Plateau (Barrett unpublished data). The majority of flights (> 40 m) in strips of vegetation were by larger open-country species such as eastern rosellas, Australian magpies and noisy miners (60%). While the presence of noisy miners did appear to deter many species from using these strips as flyways, strips of vegetation were frequently used by smaller species such as the striated thornbill, striated pardalote, yellow-faced honeyeater and white-naped honeyeater. A similar proportion of migrant species were recorded flying along transects in strips of woodland and patches of woodland (43% and 37% respectively). This supports the conclusion that birds moving through the landscape did not concentrate in strips of vegetation.

The decrease in total bird species richness in noisy miner sites included foliage gleaners such as the striated thornbill and spotted pardalote, trunk-foragers such as the white-throated treecreeper, bark and ground feeders such as the grey shrike-thrush, and nectar-feeders such as the white-naped honeyeater and eastern spinebill (see Chapter 2 for other species belonging to each feeding guild). Similarly, Catterall *et al.* (1991) found that where noisy miners occurred along edges of urban forest remnants in southeast Queensland, foliage-gleaners such as the spotted pardalote, rufous whistler and golden whistler *Pachycephala pectoralis*, aerial-feeders such as the grey fantail, and nectar-

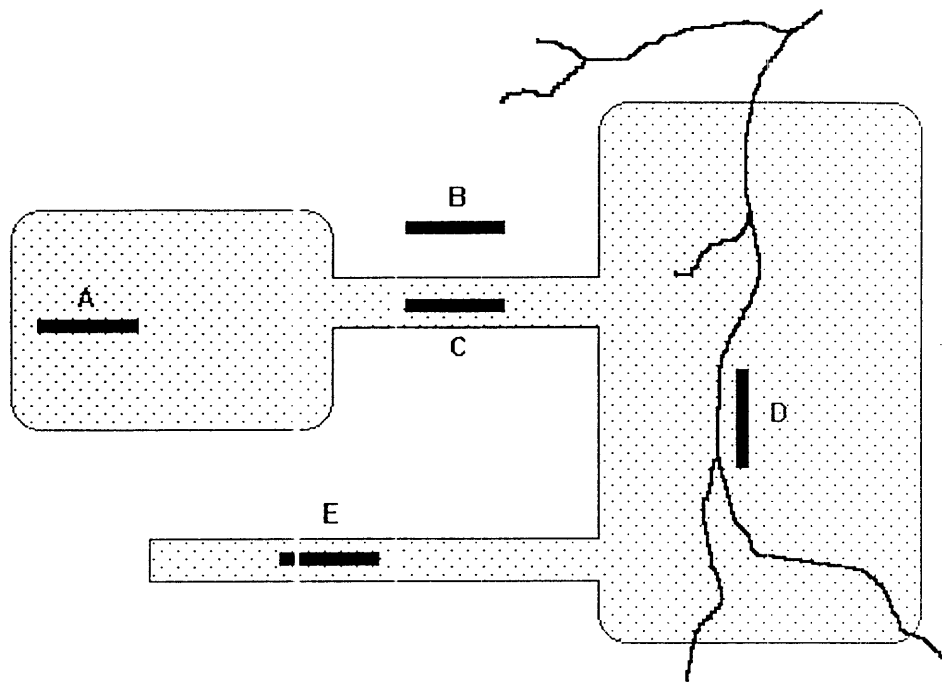
feeders such as the yellow-faced Honeyeater were less common. This result, and the results of the present study, support the suggestion by Dow (1977) that within miner territories, noisy miners exclude a wide range of other species in order to forage more generally over a range of foraging habitats. The increased abundance of open-country, hollow-nesting species such as the eastern rosella and galah in miner sites, is consistent with a previous study on the Armidale Plateau in which it was found that grassland species such as eastern rosellas and Australian Magpies (and grey butcherbirds) were the only species that regularly occurred in woodland sites where noisy miners were common (Ford 1985). Studies in other regions of eastern Australia have found that open-country, hole-nesting species such as starlings, eastern rosellas and striated pardalotes, as well as nest predators such as the Australian magpie, Torresian crow and laughing kookaburra, were common on noisy miner territories (Dow 1977; Catterall *et al.* 1991; Loyn 1985a, 1987). The present study indicates that the lower species richness within miner territories on the Armidale Plateau (see also Chapter 2; Ford 1985; Barrett *et al.* 1994), persisted throughout each season for at least two years. There was an increase in both the number of noisy miners and the total bird species richness on miner territories during spring. The increased abundance of miners in territories during spring may be due to individuals becoming more abundant and conspicuous during the breeding season. Alternatively, the population density of the noisy miners within territories may increase during spring, because of the greater effort required to defend territories during this time of increased bird species abundance on the Armidale Plateau.

The temporary establishment of separate noisy miner territories on two of the survey sites on the Armidale Plateau, suggests that some miner territories can be either mobile or transient. Both territories were established approximately 600 metres from a slightly larger territory, in an adjacent patch of stringybark woodland in which miners were present for the duration of the study. The establishment of these territories in woodland where the drooping mistletoe was flowering, suggests that such 'satellite territories' may

be established to take advantage of extra nectar availability. The temporary expansion of a noisy miner territory and resulting expulsion of bell miners *Manorina melanophrys* was described by Clarke (1984) at Eundoora, northeast of Melbourne. In his study, the miners remained for only a few days before vacating the area. The bell miners did not return, resulting in an influx of other bird species.

Figure 3.1: Positioning of strip transects in patches of woodland (A), grassland (B), corridors (C), along water-courses within patches (D) and in peninsulas (E). Transects are all 60 m wide and of variable length.









-  Wood and
-  Grass and
-  Transect
-  Water course

Figure 3.2: The mean number of bird species per site ( $\pm$  SE) in different sized woodland patches during spring, summer, autumn and winter. Data for the two years were combined and sites on noisy miner territories excluded. Woodland size categories are; > 400 ha (n = 12), 101-400 ha (n = 12), 21-100 ha (n = 15), 6-20 ha (n = 5) and < 6 ha (n = 6). When all seasons were included in the analysis, there was no significant difference between patch sizes. There was, however, a difference when spring and autumn were combined and analysed separately ( $F = 3.0$ , d.f. = 4, 45,  $p < 0.05$ ). There was a significant seasonal effect with the number of species being greater during spring than during winter in patches that were between 101 and 400 ha ( $F = 10.7$ , d.f. = 3, 135,  $p < 0.01$ , compare a and b). There was also a significant interaction between area and season (MANOVA; Wilks' Lambda = 0.56, d.f. = 12, 114,  $p = 0.01$ ).

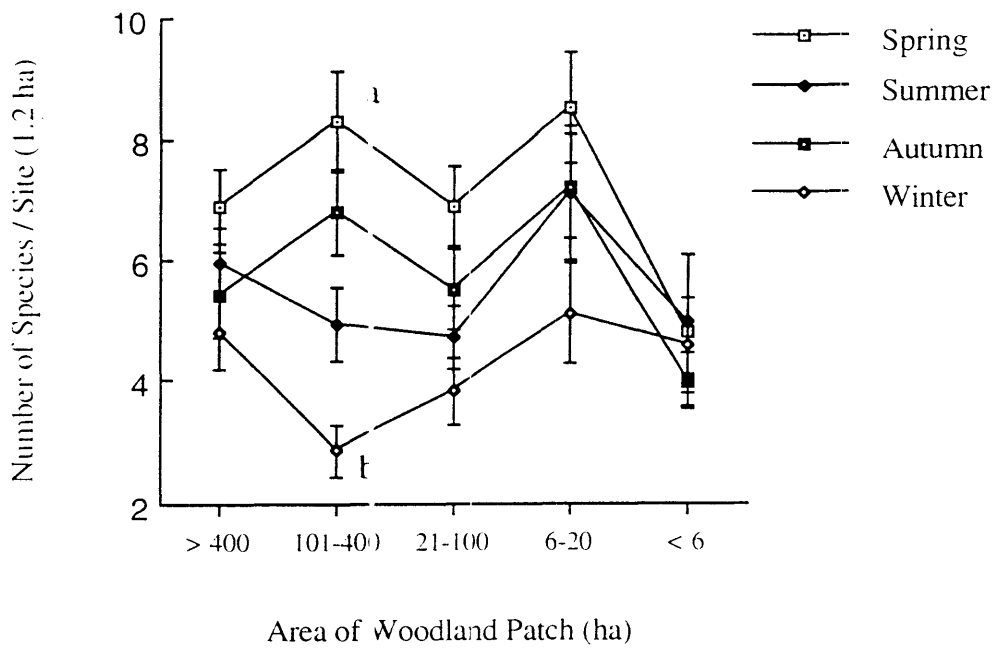
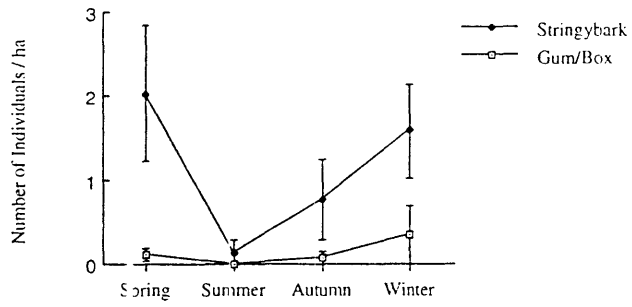
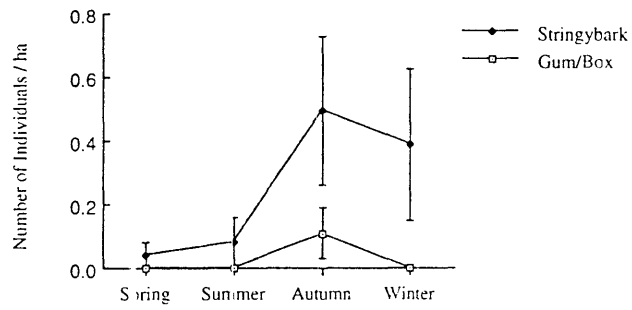


Figure 3.3: Seasonal variation in the number of white-naped honeyeaters (a), yellow-faced honeyeaters (b), noisy miners (c) and fuscous honeyeaters (d), in gum-box woodland and stringybark woodland (means  $\pm$  SE). Data were combined for the two years (n = 6 transects in both Gum / Box woodland and Stringybark woodland, 8 surveys over two years).

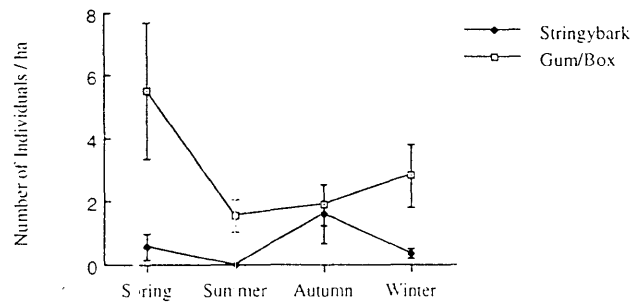
a) White-naped Honeyeater



b) Yellow-faced Honeyeater



c) Noisy Miner



d) Fuscous Honeyeater

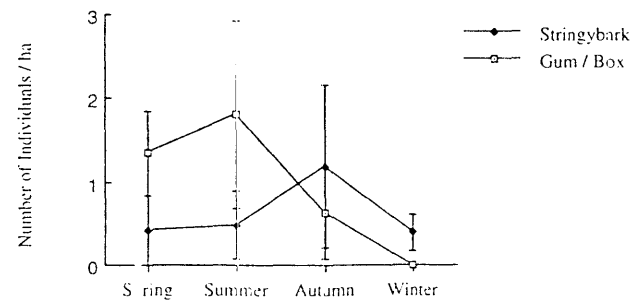
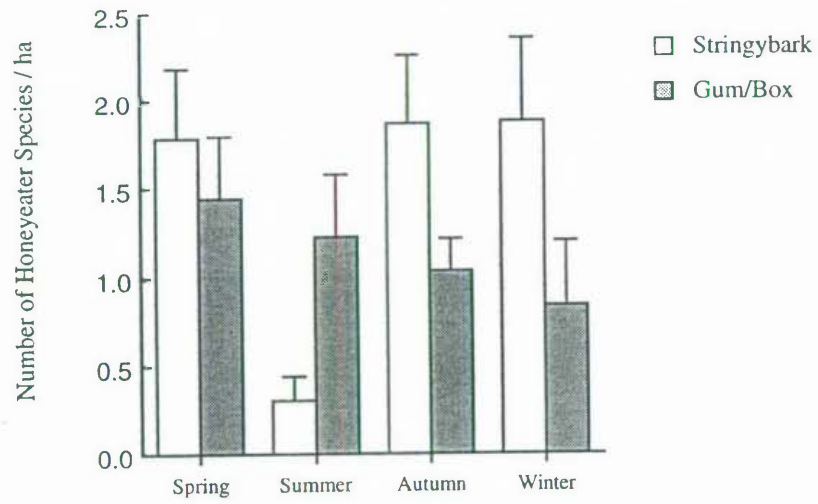


Figure 3.4: Seasonal variation in the number of honeyeater species per hectare in gum-box woodland and stringybark woodland, during the first year (a) and second year (b) of the surveys. There was a significant interaction between woodland type and season (repeated measures ANOVA,  $n = 6$ ,  $F = 4.5$ ,  $d.f. = 3, 30$ ,  $G-G < 0.05$ ). SNK tests showed no significant differences between the means.

a) 1990 - 1991



b) 1991 - 1992

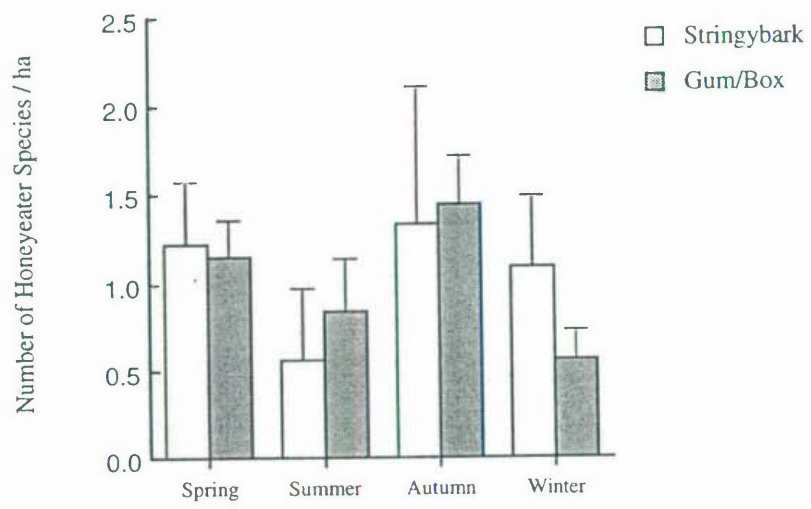


Figure 3.5: Seasonal variation in the mean ( $\pm$  SE) number of bird species (a) and the number of birds (b) on transects in woodland patches ( $n = 7$ ), along water-courses within woodland patches (Water-C.  $n = 6$ ), in woodland corridors ( $n = 3$ ), peninsulas ( $n = 3$ ) and on grassland transects ( $n = 4$ ). Transects were surveyed each season for two years and totals were combined. There was a significant variation in the number of bird species per hectare ( $F = 16.4$ , d.f. = 4, 17,  $p < 0.01$ ) and birds per hectare (all species combined,  $F = 13.4$ , d.f. = 4, 17,  $p < 0.01$ ) between transects in patches, along water-courses, corridors, peninsulas and in grassland. During summer, autumn and winter the number of bird species was significantly greater on transects in patches than those in grassland ( $p < 0.05$ , SNK for adjusted means). During summer the number of species was also greater along water-courses than in grassland ( $p < 0.05$ ; SNK for adjusted means, compare \*).



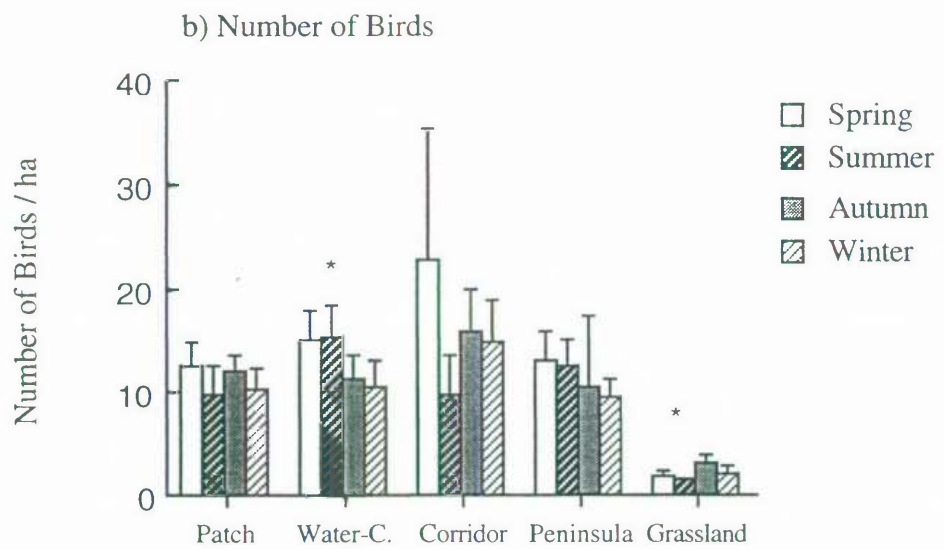
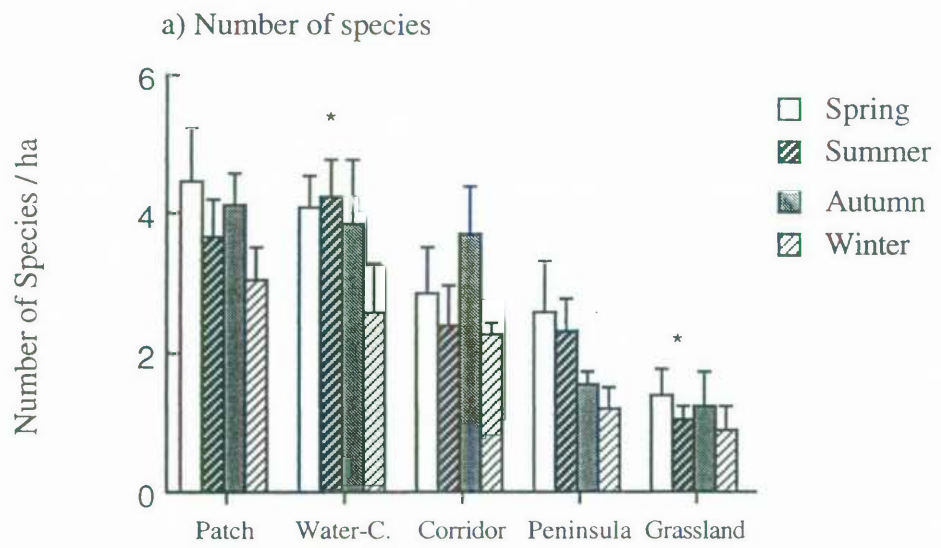


Figure 3.6: The total number flights (> 40 m, along and across transect combined) in woodland patches (n = 7), along water-courses within woodland patches (n = 6), in woodland corridors (n = 3), peninsulas (n = 3) and on grassland transects (n = 4). Flights were totalled over two years (once each season - 8 surveys in total). The number of flights was significantly greater in corridors than in grassland ( $F = 4.7$ , d.f. = 4, 17,  $p = 0.01$ , seasons combined, SNK test;  $q = 5.9$ ,  $r = 5$ , d.f. = 17,  $p < 0.05$ ).

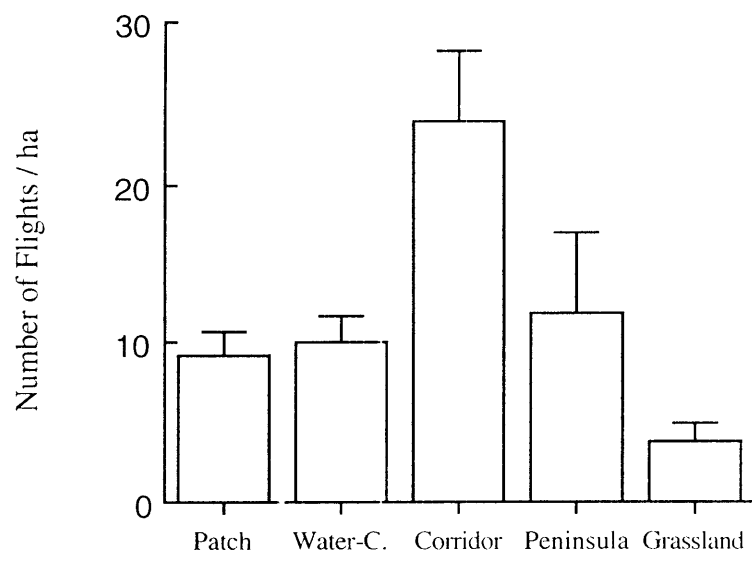


Figure 3.7: The number of flights (> 40 m) along and across transects (means  $\pm$  SE), during spring (a), summer (b), autumn (c) and winter (d), in woodland patches (n = 7), along water-courses within woodland patches (n = 6), in corridors (n = 3), peninsulas (n = 3) and on grassland transects (n = 4). There was significant variation in the number of flights per hectare (greater than 40 m in length) on transects in patches compared with those along water-courses (Water-C.), corridors, peninsulas and in grassland ( $F = 4.2$ , d.f. = 4, 35,  $p < 0.01$ ). There was also a significant interaction between season and whether the flight was along or across the transect ( $F = 3.5$ , d.f. = 3, 105,  $G-G < 0.05$ ).

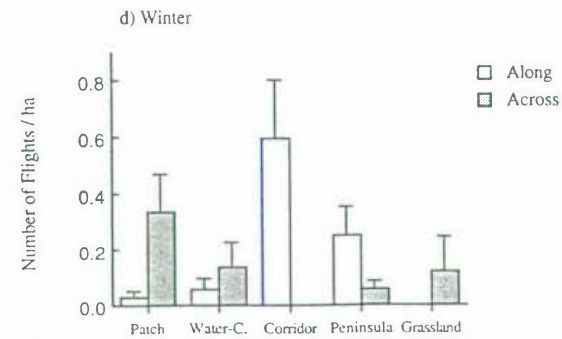
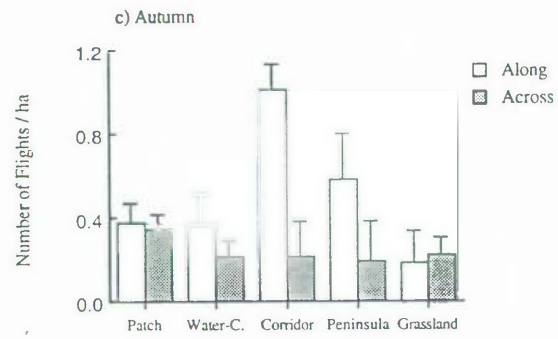
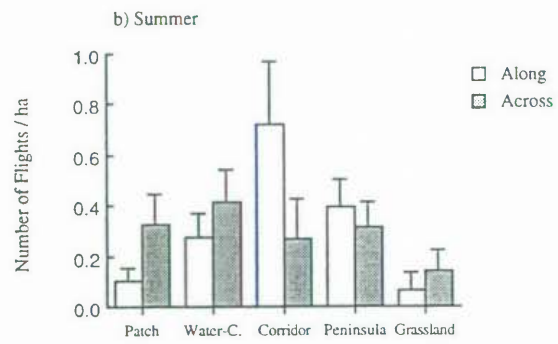
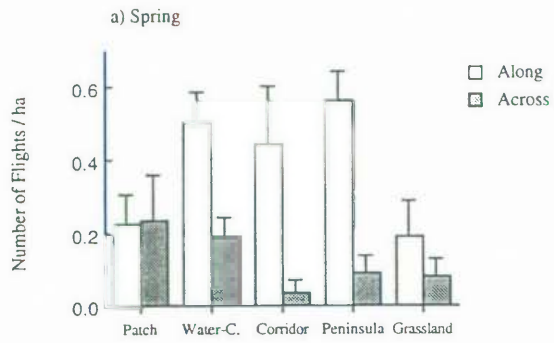


Figure 3.8: The mean number of bird species per hectare ( $\pm$  SE) after two 20 minute surveys, in control sites and sites in which noisy miner territories were temporarily established during winter 1991 (2 transects, 2 surveys;  $n = 4$ ). The number of species was significantly lower on transects after noisy miners arrived ( $X^2 = 10.9$ , d.f. = 2,  $p < 0.01$ ), there was no significant change in the control sites.

