CHAPTER 2: FACTORS AFFECTING WOODLAND BIRD ABUNDANCE AND DISTRIBUTION IN A FRAGMENTED RURAL LANDSCAPE

2.1 INTRODUCTION

The landscape on the New England Tablelands is predominantly pastoral, with sheep and cattle grazing as the primary land-use. Presently, more than half of the woodland and forest has been cleared for pasture (Smith and Turvey 1977). Like many other rural landscapes throughout Australia a large portion of the bird species are threatened by current land management practices (Ford *et al.* 1985, 1995; Howe 1986; see also Chapter 4). Howe (1984, 1986) found that bird species richness decreased in forest 'islands' (0.5 to 7 ha) when compared with extensive forest areas (> 500 km ^{2}) on the New England Tablelands. He also concluded that the area of the forest patch was the primary predictor of bird species richness, and demonstrated that open-country species replaced forest-dependent species in the forest 'islands'. In the same study Howe showed that there was a greater species turnover in plots within the extensive forest areas compared with the forest 'islands', concluding that forest remnants on the New England Tablelands were behaving as "oceanic islands" as predicted by equilibrium biogeography (MacArthur and Wilson 1963, 1967). As a result of his study, Howe (1986) suggested that large forest 'islands' or areas of continuous forest be given priority as conservation areas.

While Howe's work indicates that area undoubtedly influences bird assemblages on the New England Tablelands, it should be noted that the forest remnants were chosen to minimise habitat differences other than area of remnant and the degree of isolation. As a result, the influence of environmental variables other than area was reduced. It should also be noted that Howe's (1984) result was based on repeat surveys, and that within 'island' sites, he surveyed a proportionally larger area in the larger islands (which ranged in size from 0.6 ha to 6 ha). It is to be expected that more species will be recorded in sites where more time is spent searching (Connor and McCoy 1979). Verner (1986) has argued that in order to understand better the influence of the area of remnant on species richness, surveys should be done within areas of equal size across a range of remnant patch sizes. Where this has been done, a number of environmental features have been identified as being more important than area as predictors of bird species richness and abundance (Lynch and Wigham 1984; Freemark and Merriam 1986; Blake and Karr 1987; see also Chapter 1).

Habitat degradation is likely to have an important influence on bird assemblages of the Armidale Plateau. Ford and Bell (1981) found that bird species richness was significantly reduced in woodland patches that were affected by eucalypt dieback, partly because these sites tended to be occupied by noisy miners *Manorina melanocephala* (see also Chapter 4). These authors also noted that nest predators were more common in sites affected by dieback, which may further contribute to the loss of bird species in these sites. Davidson and Davidson (1992) have argued that understorey vegetation provides habitat for a range of native fauna, and its protection throughout the landscape is necessary for sustainable land management on the New England Tablelands. Grazing pressure by sheep and cattle is more intense in smaller woodland remnants, resulting in a removal of the understorey vegetation and a degradation of the ground ecosystem (Ford and Bell 1981; Morgan 1981; Ford 1993; Ford *et al.* 1995). Dunkerley (1989) demonstrated that understorey vegetation provides a food source for honeyeater species on the New England Tablelands. Davidson and Davidson (1992) and Stokes (in prep.) have also commented on the importance of large trees with nesting hollows, as well as the importance of fallen timber as habitat for a range of native species. Both these habitat resources are threatened by timber harvesting on the Armidale Plateau (Wall and Reid 1993).

If the majority of bird species are to be conserved in a disturbed landscape, then interactions between birds and their environment must be understood in the context of existing land management practices. Ecological surveys combined with rapid habitat assessments are often used as a cost-effective way to investigate relationships between birds and their environment (Braithwaite 1991). Regression models derived from survey data have been used by numerous authors to identify habitat features that are likely to influence bird assemblages (see methods section for references). In the present study, birds were surveyed and 35 environmental features were measured in 294 woodland sites throughout the New England Tablelands. The environmental features that were most highly correlated with specific bird groups were identified and regression models constructed. This is an observational study, the aim of which is to identify habitat features that are positively or negatively correlated with bird abundance. The survey sites were chosen to represent severely degraded, through to relatively undisturbed woodland. Extensively wooded areas, as well as small patches that were isolated from other woodland were chosen. The sampling procedure was further stratified with respect to woodland type, presence or absence of noisy miners and presence or absence of understorey vegetation. The environmental features which were most frequently selected as predictors are discussed in relation to specific bird groups, and management guidelines for maintaining bird species richness on the Armidale Plateau are proposed.

2.2 STUDY AREA

The New England Tablelands were first settled by graziers in the 1830s. Though widespread clearing of woodland had stopped by the 1930s, woodland clearing continues today (Davidson and Davidson 1992; Curtis *et al.* in press). The study site, which will be referred to as the Armidale plateau, is the upland area within a 60 km radius of Armidale, on the New England Tablelands (151° 40' E, 30° 32' N). The

Armidale Plateau is intensively grazed. Remaining tree cover is less than 20% and mostly restricted to rocky, sloping country on poor soils (Figure 2.1). Most of this tree cover is woodland (< 30% canopy cover) rather than open forest (30 to 70% canopy cover). Conventionally this landscape has been referred to as fragmented, which suggests discrete patches of continuous woodland surrounded by treeless grassland. However, most of the remnant woodland occurs in diffuse fragments that are surrounded by scattered trees, a landscape that is more accurately described as variegated (McIntyre and Barrett 1992).

About half of the woodland on the Armidale Plateau is affected by eucalypt dieback (Jones *et al.* 1990). Dieback, caused by repeated defoliation due to leaf-eating insects, is likely to be the result of intensified grazing practices since the 1950s (Lowman and Heatwole 1992). Fragmentation of the woodland habitat may also have caused dieback, by reducing the number of predators and parasites of herbivorous insects in smaller isolated remnants (Recher pers. comm.; see also Davidson and Davidson 1992). 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 Blakely's red gum *E. blakelyi* and yellow box *E. melliodora* association, restricted to the richer soils at lower altitudes. It is the flatter country at lower altitudes that has been most intensively cleared and modified by agricultural use. 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*, blackthorn *Bursaria spinosa*, sifting bush *Cassinia quinquefaria* and 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).

2.3 METHODS

Birds were surveyed and the habitat described in 294 woodland sites on the Armidale Plateau (Figure 2.1). On 63 of these sites bird surveys were repeated each season for two years (a total of eight 20 minute surveys). Seasons were defined as follows; Summer - December, January, February, Autumn - March, April, May, Winter - June, July, August, Spring - September, October, November. The survey sites were chosen using a stratified sampling technique (Eberhardt and Thomas 1991), the aim of which was to capture the gross variations in woodland type and bird community on the Armidale Plateau. Sites chosen included roughly equal portions of large and small remnants, isolated and non-isolated remnants, and degraded and healthy remnants. In hindsight, small remnants of gum-box woodland were under-represented in the present study.

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. An area of woodland was considered a separate patch if for the most part, it was more than 50 m from an adjacent area of woodland. In many cases, patches were connected by strips of woodland which were defined as being less than 100 m wide and more than 300 m long. In most other cases patches were connected by scattered trees.

2.3.1 Bird Survey

All birds seen or heard within 30 metres either side of the observer were recorded along a 200 metre strip transect (area of strip transect is 1.2 ha). Transect counts are a widely used and accepted survey procedure in Australia (for a review of methods see Pyke and

Recher 1984; Recher 1988). As described previously, the woodland habitat throughout the study area is open, so most bird species could be seen or heard comfortably within a distance of 30 metres. A transect length of 200 m was chosen because this was short enough to fit into remnants as small as 2 ha. Unfortunately, remnants that were smaller than 2 ha are under-represented in this study. Many of these smallest remnants were highly degraded patches of gum-box woodland. Trial surveys were repeated at time intervals of three, four, five and eight minutes per 50 m of transect. The largest increase in the mean number of species recorded along a 200 m transect occurred between four and five minutes per 50 m, so the rate of movement along the transect was fixed at five minutes per 50 m. Spending more than 20 minutes surveying a 200 m transect increases the risk of double counting individuals. All surveys were carried out in the morning.

The total number of species, total number of individuals and Simpson's diversity index (Begon *et al.* 1990) were calculated for each of the 294 woodland sites. In Table 2.1, bird species are combined into groups on the basis of taxonomy, abundance, foraging guilds, habitat preference and nesting behaviour (Frith 1976; Ford *et al.* 1986). The number of sites out of 294 in which each bird species was present is also given. Common species were defined as occurring on at least 22% of the sites (more than 65 of the 294 sites) and uncommon species defined as occurring on less than 4% of sites (fewer than 10 of the 294 sites). Species with low population density were defined as having fewer than 1.3 individuals per hectare in sites where they occurred. Species with high population density were defined as having greater than 2.5 individuals per hectare. The seasonal surveys described above were used to identify species as resident species, summer visitors or winter visitors. These categories were confirmed by previous research on the Armidale Plateau (Ford *et al.* 1986; Dunkerley 1989; Ford *et al.* 1990; Ford and McFarland 1991). Although particular bird species were regarded as summer or winter visitors, it should be noted that in some cases these species were partial migrants, with some individuals remaining for most of the year. There was also an overlap between the arrival and departure of summer and winter visitors, resulting in both bird groups being present during the single survey in spring 1990. Scientific names for bird species are given in Table 2.1. For bird species not included in the analysis, scientific names are given when the species is first mentioned in the text.

2.3.2 Environmental Variables

Thirty-five environmental features were described for each of the 294 sites in which birds were surveyed. Many of these variables were similar to those used by Lynch and Wigham (1984) and Robbins *et al.* (1989). Mean values for each of the 35 features, as well as the transformation applied prior to standardization and analysis are given in Table 2.2.

Trees, Shrubs and Herb Vegetation

The large number of sites and the limited time spent collecting bird survey data at each site (20 minutes), meant that it was neither possible nor appropriate to do more than a rapid assessment of vegetation. At each site the 200 m transect (1.2 ha) was divided into four 50 m sections (each 50 m long and 60 m wide), and vegetation features were estimated separately for each quarter. An average value was then calculated for each site. Up to ten different observers were used to collect habitat data for all 294 sites. Problems associated with observer bias (Gotfryd and Hansell 1985) were minimised by taking the following precautions; observer estimates were calibrated before and during the habitat survey; observers worked in pairs, estimating environmental variables independently and comparing estimates; and pairs of observers were randomly assigned to sites. Other researchers have shown that rapid habitat assessments are adequate for describing relationships between birds and their environment (Howe 1984; Arnold 1988; Austin 1991; Braithwaite 1991; Woinarski and Braithwaite 1993).

A periscope, over which a grid had been placed, was used to calibrate visual estimates of percentage canopy cover (> 8 m) and sub-canopy cover (2 - 8 m). Visual estimates were also used for percentage shrub cover (< 2 m), percentage herb cover and dominant height of the herb vegetation. A clinometer was used to estimate the dominant canopy height and the height of the tallest tree. Tree species were identified and counted on each 50 m section of the strip transect, the four counts were then averaged for the site. Shrub species were also identified and the percent shrub cover that was due to *Acacia* spp. was estimated.

For the following environmental variables, numerical categories were used to reduce field description time and improve consistency: The average tree diameter was categorised as 10 cm if all trees were < 20 cm in diameter, 15 cm if all tree diameters were < 30 cm but some were > 20 cm, 25 cm if some were > 30 cm and many were > 20 cm, 35 cm if some were > 40 cm and many were > 30 cm, and 45 cm if many were > 40 cm in diameter. Fallen trees or branches greater than four metres in length and > 20 cm in diameter (hereafter referred to as logs) was categorised as; 0 if no logs were present, 1.5 if 1 - 2 logs were present, 3.5 if 3 - 4 logs present, 6 if 5 - 8 logs present and 10 if more than 8 logs were present. Two species of mistletoe were common throughout the Armidale Plateau; the drooping mistletoe *Amyema pendulum* found mostly on stringybark trees, and the box mistletoe *A. miquelii* found mostly on gum and box trees. The counts for each species of mistletoe was categorised as 0 if no mistletoes were present, 2.5 if 1 - 5 mistletoes present, 8 if 6 - 10 mistletoes present, 15 if 11 - 20 mistletoes present and 25 if there were more than 20 mistletoes were present. Cut stumps was categorised as 0 if there were no cut stumps, 1.5 if 1 - 2 stumps were present, 3.5 if 3 - 4 stumps were present, 6.5 if 5 - 8 stumps were present and 10 if there were more than 8 cut stumps present. The extent of dieback on each 50 m section of the transect was estimated using a modified method from Nadolny (1984). Dieback was categorised as 0% if the tree appeared healthy, 25% for trees with dead twigs on primary crown and some epicormic growth, 50% for moderate levels of dieback throughout, 75% for trees with predominantly epicormic growth and most of the twigs on the higher crown apparently dead, and 100% for severe dieback where many trees are defoliated and where foliage does occur it is mostly epicormic growth restricted to the main trunk.

Slope was categorised as 0 if flat, 1 if gentle and regular, 2 if moderately sloping or gently undulating, 3 if steep or undulating and 4 if very steep and undulating. Numerical categories for each environmental variable were estimated for each 50 m quarter of the transect and an average calculated for each site.

Landscape Variables

Altitude (m) and area (ha) of each woodland patch were calculated using 1:25 000 topographic maps and aerial photos Distance (km) from the centre of the transect to the nearest town, farm buildings, road, permanent water and water-course were also recorded. In most instances permanent water was a farm dam, although it was sometimes a river. Water-courses may have had no water present when the bird survey was conducted. The distance (km) from the centre of the transect to the woodland edge was also recorded. The majority of transects started from, and were perpendicular to, the edge of the woodland patch.

Area of Woodland Patch

The effect of area of woodland patch on the number of bird species was investigated in greater detail by dividing the 294 sites into five size categories. These categories were further divided into sites with noisy miners (on which more than one noisy miner was recorded during the single 20 minute survey) and sites without noisy miners (on which one or no miners were recorded). The categories are as follows: > 400 ha (miner site n = 5, non-miner site n = 72), 101 - 400 ha (miner site n = 5, non-miner site n = 74), 21 - 100 ha (miner site n = 17, non-miner site n = 65), 6 - 20 ha (miner site n = 15, non-miner site n = 20), < 5 ha (miner site n = 7, non-miner site n = 14) and strips of woodland (defined as being less than 100 m wide and more than 300 m long, miner site n = 5, non-miner site n = 21). The number of bird species was recorded for each of these categories. Means were compared using the Kruskal-Wallis test (Sokal and Rohlf 1981) and *a posteriori* tests of mean ranks were carried out using Dunn's multiple comparison technique (Neave and Worthington 1988).

Area effects on the number of bird species were also investigated separately for both stringybark and gum-box woodland. For each of the five size categories described above, sites in which the proportion of stringybark to gum-box woodland was greater than 0.7 were analysed separately (> 400 ha, n = 16; 101 - 400 ha, n = 35; 21 - 100 ha, n = 37; 6 - 20 ha, n = 15; < 6 ha, n = 11). These stringybark sites were then compared with sites in which the proportion of gum-box to stringybark woodland was greater than 0.7 (> 400 ha, n = 28; 101 - 400 ha, n = 19; 21 - 100 ha, n = 27; 6 - 20 ha, n = 15; < 6 ha, n = 8). Noisy miner sites were included in this analysis. Data were log-transformed and the mean number of species compared using analysis of variance.

The cumulative number of bird species was also calculated, using the surveys which were repeated each season for two years (eight 20 minute surveys). Sites with noisy miners were included in the analysis. These were sites in which noisy miners were present in at least five of the eight surveys. The 63 sites in which surveys were repeated (subsequent to the initial spring 990 survey) were divided into the same five size categories described above: > 400 ha (n = 12, includes no miner sites), 101 - 400 ha (n = 13, includes 1 miner site), 21 - 100 ha (n = 24, includes 7 miner sites); 6 - 20 ha (n = 6, includes 3 miner sites) and < 6 ha (n = 8, includes 2 miner sites).

Derived Environmental Variables

The number of trees counted for each tree species was converted to a proportion of gum, box, gum-box and stringybark. Vegetation biomass was calculated by adding estimates of percent canopy cover, sub-canopy cover and percent shrub cover. An isolation index was calculated as a function of the following three variables; the percent woodland cover in a 4 km ^{2} area immediately surrounding the patch, the number of woodland patches in the same 4 km ^{2} area, and the 'connectivity' of the patch. The number of surrounding woodland patches was coded as 0 for none, 1 for 1 - 3 surrounding patches, 2 for 4 - 6 patches, 3 for 7 - 9 patches and 4 for > 10 surrounding patches. Connectivity was coded as 0 if the patch was not physically connected to any other wooded areas, 1 if it was connected to woodland patches < 100 ha in size, 2 if poorly connected to patches > 100 ha in size, 3 if there was a continuous, broad connection to a patch > 100 ha in size and 4 if there was more than one good connection to patches > 100 ha in size. For each site, these three measurements of isolation were standardized by dividing each score by its maximum, summed, added to one and then inverted to produce an isolation index (I).

The formula for estimating isolation is as follows:

$$\text{Isolation} = \frac{1}{(1 + \%\ \text{surrounding woodland} + \#\ \text{woodland patches} + \#\ \text{connections})}$$

For an isolated patch with no surrounding woodland I = 1.0. The least isolated patch of the 294 sites surveyed (with an isolation index value of I = 0.36) was in an 11 ha patch that had no connections to other woodland areas. However, there were greater than ten woodland patches, covering 51% of the 4 km ^{2} immediately surrounding the patch. For sites in patches of woodland that were greater than 400 ha, the woodland cover in a 4 km ^{2} band immediately outside an area of 400 ha area, centred on the survey site, was used for calculating the isolation index. The maximum surrounding woodland cover across all sites was 65%.

2.4 ANALYSIS

There are differing views on which multivariate analysis should be used to build predictive models from a large variable data set, such as the one presented here. It is generally accepted, however, that more than one kind of analysis is appropriate, and that the methods chosen should include both parametric and non-parametric, as well as linear and non-linear methods (Maurer 1986; James and McCulloch 1990). The primary reason for using more than one analysis is to look for consistency between the predictive models derived using the different approaches. Although it has been recommended that predictive models of bird abundance should include species interactions, such as the presence or absence of noisy miners (Mac Nally 1990a), it was decided to construct the models using only the environmental variables. This is because the aim of this study was not to maximise the amount of variation that was accounted for by each model. Rather this study was concerned with identifying environmental features that were most likely to be influencing the distribution and abundance of bird groups.

Three different kinds of analyses were used to investigate correlations between the derived bird groups and the environmental variables. Two of these used regression techniques and are parametric methods, that construct linear regression models and have relatively strict statistical assumptions (James and McCulloch 1990). Evaluation of these assumptions led to transformation of the habitat data (for transformations see Table 2.2) to improve normality, linearity, and homoscedasticity of the residuals (Tabachnick and Fidell 1989). After transformation, habitat data were standardized to put all environmental variables on the same scale, by subtracting the mean and dividing by the standard deviation (Sokal and Rohlf 1981). Counts of individual bird species at each of the 294 sites were also standardized by subtracting the mean from each count and dividing this difference by the standard deviation. The standardized values were then summed for each bird group This reduced the chance that the distribution and abundance of groups of bird species would be dominated by a few common species, and reduced the likelihood that statistical assumptions would be violated (Sokal and Rohlf 1981, Okasen 1983, Wilkinson *et al.* 1992, Clarke 1993). In ecological terms, this standardizing of the bird count data means that counts for species occurring at low population density in a few sites are slightly increased, relative to those species occurring at high population density throughout the study area. For example, standardizing the data means that a site which has, say two hooded robins (an uncommon species) is given greater importance than a site which has two scarlet robins (a species which occurs more commonly throughout the study area). As a result of standardization, the abundance and distribution of a particular guild of species is more likely to reflect all of the species included in the guild rather than a handful of common species.

Rather than being standardized, counts of total number of bird species and total number of individuals were square-root transformed and estimates of diversity using Simpson's diversity index were log-transformed. Data from the repeated surveys on 63 sites were analysed using standard multiple regression only (principal components analysis and HMDS were not performed using this smaller data set). Each species was scored 0 - 8, depending upon the frequency of occurrence over the eight seasonal surveys. These scores for each species were standardized by subtracting the mean and dividing by the standard deviation, and then summed for each bird group. Missing data were replaced by means (Tabachnick and Fidell 1989). All of the regression models presented, as well as the variables within the models, are significant ($p < 0.05$).

The third statistical method - hybrid multidimensional scaling (HMDS), is a non-parametric, non-linear technique that does not require data transformation prior to analysis (James and McCulloch 1990). However, the data were standardized as described below.

2.4.1 Regression Using Principal Components

The 35 environmental variables were reduced to a smaller number of principal components. The principal components analysis was performed with varimax rotation using the correlation matrix, through SYSTAT (Wilkinson *et al.* 1992). The principal components were then regressed against the bird groups using stepwise regression

(Tabachnick and Fidell 1989). This method has the advantage that all of the environmental variables are used to construct the principal components, so *a priori* decisions about which environmental variables are most important are not necessary (Maurer 1986). Principal components analysis is robust against violations of statistical assumptions (James and McCulloch 1990) producing predictive models that are general rather than precise (Maurer 1986). Principal components analysis has been used to describe the relationship between birds and habitat by James (1971), Blake and Karr (1987) and Arnold (1988).

2.4.2 Standard Multiple Regression

Single 20 minute survey (spring 1990)

The 294 sites were randomly divided in half (two data sets each of 147 sites). One half, the 'estimation' sample was used to generate the regression models and the other half, the 'validation' sample, was used to test the models (Wittink 1988). The 35 environmental variables in the estimation sample, were clustered into nine relatively uncorrelated groups (Appendix 1) using hierarchical clustering, with the euclidean distance metric and complete linkage for amalgamation (Wilkinson *et al.* 1992). Each bird group was then correlated with all 35 environmental variables and a single variable that was most highly correlated was selected from each of the nine clusters. This method for reducing the number of variables before constructing the model is recommended by Henderson and Velleman (1981).

Predictive models were then constructed from these nine variables using standard multiple regression, using the adjusted multiple correlation R^2(ADJ). This measure corrects for chance fluctuations that result in an overestimation of the variation explained by the regression (Tabachnick and Fidell 1989). Initially, all nine variables were entered into the model and those with the highest p values and lowest tolerance levels were eliminated one at a time. The final models showed only significant variables ($p < 0.05$) that are uncorrelated ($T > 0.7$). Where the R^2(ADJ) was increased or unaffected by the removal of a variable, that variable was removed (Wilkinson *et al.* 1992).

The root average squared prediction error (RASPE) was used as a cross-validation procedure (Wittink 1988). The RASPE was calculated for both the estimation sample (the data set from which the model was built) and the validation sample, by subtracting predicted values for each bird group from actual values, and then summing the squared differences and finding the square root. This value provides a measure of the internal consistency of the regression model. A measure of the loss of information when the model is used to predict values, can be made by comparing the RASPE for the estimation sample to that for the validation sample (Maurer 1986). As an additional measure of the predictive accuracy of the models, the mean predicted value was compared with the mean of the actual values, and if the difference was significant the model was rejected ($P < 0.05$).

Repeated Measures on 63 Sites (8 surveys each season for two years)

Regression models for the repeat censuses on 63 sites were constructed using the same method as that for the single survey on 294 sites, except that all 63 sites were used to construct the model and these models were not cross-validated. As mentioned previously, each bird species was scored 0 - 8 depending on the frequency of occurrence over the eight surveys, before being standardized and summed into one or more of the bird groups (Table 2.1). These models put less emphasis on spatial variation (only 63 sites instead of 147), however, they take into account temporal variation. A combination of both spatial and temporal variation was recommended by Gaud *et al.* (1986) as the optimal survey procedure for regression modelling.

In comparison to constructing regression models using principal components, regression modelling using specific environmental variables sacrifices generality for increased precision (Maurer 1986). However this kind of analysis is not as robust to violations of statistical assumptions (James and McCulloch 1990; Nicholls 1991). Of the three methods used in this study, the regression analysis using specific environmental variables is most commonly used to investigate the relationship between birds and habitat (Hone 1991). The use of regression analysis to investigate this relationship has been demonstrated by Howe (1984), Lynch and Wigham (1984), Freemark and Merriam (1986), Braithwaite *et al.* (1989) and Cale (1990).

2.4.3 Hybrid Multidimensional Scaling (HMDS)

The validity of using parametric, linear modelling techniques for multivariate community ecology data, where there is a pattern of successive replacement of species through space, has recently been questioned (Minchin 1987; Austin 1991; Crisp and Weston 1993). Non-metric multidimensional scaling, which uses the ranks of dissimilarity measures among objects rather than the dissimilarity measures themselves, is recommended to community ecologists as a robust technique for indirect gradient analysis (Kenkel and Orloci 1986; Minchin 1987; James and McCulloch 1990; Clarke 1993). This is a non-parametric method which has only the single assumption of monotonicity (constant rate of increase). Faith *et al.* (1987) and Belbin (1989) have developed a "hybrid" multidimensional scaling technique (HMDS) that is metric (linear) for low dissimilarities and non-metric (monotonic) above a threshold value. The rationale derives from the observation that the relationship between the dissimilarity coefficient and the ecological distance is usually linear at small ecological distances, but monotonic at best when distances are high. Metric multidimensional scaling and non-metric multidimensional scaling often give results similar to those of principal components analysis, but their main disadvantage relative to principal components analysis is that the axes are not functions of the original variables. In fact, variables do not enter the analysis, only inter-object distances are used, so direct interpretations must be qualitative and subjective (James and McCulloch 1990). However, Belbin (1989) developed an algorithm (PCC), dubbed principal axis correlation, that fits a set of attributes to an ordination space using multiple linear regression. These attributes may be extrinsic (habitat data) or intrinsic (bird data used for ordination). Principal axis correlation takes each variable and finds its best-fit vector in the ordination space (vectors of maximum product-moment correlation for each variable), resulting in two pieces of information; the direction of the vector and its correlation with that direction. The correlation coefficients can be taken to indicate the significance of each vector associated with a particular environmental variable. The main advantage of HMDS is that it is able to identify non-linear relationships, is robust to violations of statistical assumptions, and is able to cope with unavoidable weaknesses of survey data including data noise and biases in sampling (Faith 1991). This method asks the question, 'Does this community pattern implicate a particular environmental factor?' In the present study, the environmental variables that are most correlated with a bird group ordination are identified for comparison with the models produced from the regression analyses.

Hybrid multidimensional scaling (HMDS) was performed using the SSH module in the PATN package (Belbin 1989). Bird species that occurred on fewer than ten of the 294 sites were excluded from the ordination. To avoid a result dominated by a few common species, counts for each bird species were standardized by subtracting the minimum value and dividing by the range The Bray-Curtis measure, a robust measure of dissimilarity, is recommended for this ordination technique (Faith *et al.* 1987). For ease of interpretation, the number of dimensions in the ordination was kept to a minimum, ranging from three to five depending upon the bird group. In all cases stress was kept below 0.21, the average being 0.17. Stress of < 0.1 corresponds to a good ordination with no real risk of drawing false inferences (Clarke 1993).

After the ordination was performed, the PCC algorithm in the PATN package (Belbin 1989) was used to fit the environmental data to the ordination space. PCC finds a straight line that best represents the 'topography' of the surface defined by the ordination for each group of bird species. The longer this vector, the greater the correlation coefficient and the more important that variable is in explaining the ordination pattern (Belbin 1991). The six most influential environmental variables were identified (those with the six highest correlation coefficients). A method similar to that described by Crisp and Weston (1993) was then used to identify whether these 'most influential' environmental variables had a positive or negative influence on the abundance of the associated bird group. The ordination axes were correlated with the standardized bird counts that were used to construct the ordination (also the bird data used for the regression models). These correlation coefficients were then compared with those from the correlation between these same ordination axes and the six 'most influential' environmental variables.

Non-metric MDS has been used to relate environmental variables to species distribution by the following researchers: Okasen (1983), Kenkel and Orloci (1986), Belbin (1991), Clarke (1993), Crisp and Weston (1993) and for birds and environmental features Mac Nally (1989 and 1990a).

2.5 RESULTS

2.5.1 Cluster Analysis of Environmental Variables

The cluster analysis based on a random sample of half of the survey sites (estimation sample, n = 147 sites) is treated here as being representative of the whole study area (Appendix 1). Spearman's rank correlation coefficients for a reduced set of 28 variables were also generated using the estimation sample (Table 2.3). A general interpretation of the relationship between the 35 environmental variables is presented in Table 2.4. In summary, large woodland patches were further from farm buildings and roads than smaller woodland patches, and so were generally less disturbed by grazing and timber removal. Smaller woodland patches tended to be affected by eucalypt dieback. However, the herbaceous vegetation was well developed and mistletoes were more common in these smaller patches. Both large and small patches were further divided by altitude. Sites at higher altitude tended to be stringybark woodland on the nutrient-poor, sloping country, whereas sites at lower altitude were more likely to be in gum-box woodland on the flatter, nutrient-rich soils. Larger trees and logs were more common at higher altitudes, while vegetation species richness and vegetation biomass were greater at lower altitudes (Table 2.4). This interpretation was based on the cluster analysis (Appendix 1), the correlation matrix (Table 2.3), the principal components analysis of the habitat data described below (Table 2.5), as well as a knowledge of the study area. In a few cases, specific correlation coefficients are not consistent with the general interpretation. This, however, is to be expected when describing patterns in a large complex data set.

2.5.2 Principal Components Analysis of Environmental Variables

Eleven principal components were extracted from the 35 environmental variables explaining 69.5 percent of the variation (Table 2.5). Sites that were positively correlated with each principal component can be summarised as follows:

PC1	A mix of tree and shrub species
PC2	Stringybark woodland
PC3	Many mistletoes
PC4	Large, tall trees
PC5	Dense shrub and sub-canopy layer
PC6	High tree density and a dense canopy layer
PC7	Large patches of woodland, away from buildings, permanent water and away from the edge of the patch
PC8	Affected by dieback
PC9	Higher altitude
PC10	An extensive, tall herb layer
PC11	Sloping country close to water-courses

2.5.3 Predictive Models

The predictive models from the regression analysis and most likely predictors selected using the HMDS are presented in Appendices 2 to 5. A list of predictors selected using the different analyses for each bird group is presented in Table 2.6. An additional loose copy of Table 2.6 is provided for easy reference while reading the results section of this chapter. The relative frequency with which each environmental variable was selected as a predictor is given in Table 2.7. Table 2.7 also indicates the percentage occurrence of each of the environmental variables within a given analysis. Where a group of environmental variables were correlated, one variable was selected in preference to another. As a result some environmental variables were 'overlooked' by particular types of analyses, confirming the view that more than one kind of multivariate analysis is appropriate when analysing ecological survey data (Maurer 1986; James and McCulloch 1990). The stepwise regression models based on the principal components analysis (Table 2.7 - PCA), were the only analyses to emphasise the importance of large trees and tree density. The standard multiple regression based on a single spring survey (SMR - Spring), as well as the HMDS, emphasised the importance of logs rather than large trees. The standard multiple regression based on seasonal surveys emphasised the importance of area of woodland patch, rather than large trees or logs. Generally, the regression models based on the principal components analysis and the single spring survey selected a broad range of predictors. While this was also true for the HMDS, this analysis focused on altitude and woodland type as being of primary importance as predictors of bird species abundance. The standard multiple regression based on seasonal surveys (SMR - Seasonal) was the most conservative type of analysis with only three environmental variables being of overriding importance; dieback, area of woodland patch and the percent of *Acacia* spp. understorey (Table 2.7).

Despite these differences, the results from the different statistical analyses were generally consistent, producing a similar subset of environmental predictors for each bird group. The consistency between the regression models and the predictors chosen using the HMDS analysis, indicates that even if statistical assumptions were not strictly met, the regression models appear to be valid as predictive tools. This is further supported by the RASPE values, as they did not differ significantly between the estimation and validation samples. An overall summary of the relationship between the different bird groups and the environmental features is presented in Table 2.4. As with the summary of the relationship between the environmental variables (also presented in Table 2.4), there are some inconsistencies between the general summary and the specific models.

A degree of caution is recommended when interpreting the results from this analysis. The R^2 (ADJ) values associated with the regression models from a single survey are low, indicating that these linear models only capture a small part of the overall variation in the distribution of each bird group (in most cases < 20%, Appendix 3). The R^2 (ADJ) values were larger for regression models from repeated surveys on fewer sites, that is temporal, rather than spatial replication (Appendix 4). The low R^2 (ADJ) values associated with this analysis are partly due to the high probability that particular bird species were not recorded during a single visit. The large number of survey sites (294 sites for single survey), high variability in this ecosystem, and likelihood of non-linear relationships also contribute to the low R^2 (ADJ) values (Maurer 1986; Wittink 1988). As mentioned previously, the lack of inclusion of species interactions in the models also explains the low R^2 (ADJ) values (Mac Nally 1990a). Also, the relationship between bird groups and environmental variables may have been stronger if environmental variables had been measured on a larger scale (Wiens 1989). Notwithstanding these considerations, it is likely that there is a large element of randomness and chance in the distribution of woodland birds on the Armidale Plateau. Even in continuous habitat, not all of the suitable habitat may be occupied. A bird species may become extinct in one patch due to stochastic events, but remain in another patch of similar habitat. Similarly, by chance alone areas of suitable habitat will not be visited as a species declines in abundance. It is also possible that the majority of species are quite general in their habitat requirements and able to use most of the woodland habitat to some degree. If this is the case, then as has been suggested by Maurer (1986), environmental factors that explain only a small part of the variation in the distribution of a group of species may still be crucial to the well-being of those birds.

2.5.4 Regression Models for Bird Groups

The procedure that is usually followed in the literature, is to discuss individual models separately. However, due to the large number of models, the results from the different analyses will be presented collectively for each bird group. For details of the variable coefficients, model constants and R^2 (ADJ) values see Appendices 2 to 5.

Number of bird species and Simpson's Diversity Index

Bird species richness was influenced by a variety of environmental variables and the R^2 (ADJ) values from the regression models varied from 0.09 to 0.32 (Table 2.6, Appendices 2 to 4). Both the number of bird species and Simpson's diversity index showed a similar pattern of response to the environmental features. There were more bird species in stringybark woodland at higher altitude, where large trees and mistletoes were present (Table 2.6). Although the correlation was not significant in the present study (Table 2.3), large trees tend to have more mistletoes present (see also Loyn 1985b). Large trees are also likely to have more nest-hollows (Mackowski 1984; Stokes in prep.). Greater bird species richness was also associated with logs, and sites where there was a greater percentage of herb cover and greater vegetation biomass (particularly canopy cover). Bird species richness was less in sites affected by dieback and at sites in larger woodland patches that tended to be further from permanent water, such as farm dams. Bird species richness also decreased in sites further from water-courses and in sloping country. There is some inconsistency in this last statement, with water-courses tending to be in sloping country (Table 2.3). It appears that while water-courses attract a greater number of bird species, the greater herb cover associated with the flatter, richer soils also attracts bird species. Logs were associated with larger, less isolated woodland patches, in sites away from farm buildings and roads (Tables 2.3 and 2.4). The association between logs and greater bird species richness is probably due to the presence of large trees and increased canopy cover, where logs tend to be found. It is also possible that there is reduced grazing pressure in the immediate vicinity of the logs, as indicated by the greater percentage of *Acacia* spp. understorey at sites where logs were present. Fallen timber acts as a physical obstacle to livestock, allowing leaf-litter and soil to build up and shrubs to regenerate. Stock graze the nitrogen-rich *Acacia* spp. preferentially, so in sites where grazing has been continuous and heavy there is less acacia present (Scougall *et al.* 1993). The number of logs decreased in sites closer to farm buildings and roads, probably as a result of the removal of fallen timber by woodcutters. Although the relationship between bird species richness and the environmental variables is complex, generally bird species richness per unit area was greatest in small patches of woodland at higher altitudes (Table 2.4).

Number of Species and Area

Although the multivariate analyses indicate that bird species richness per site is greater in smaller woodland patches (Table 2.6), the number of bird species was not significantly correlated with the area of the woodland patch ($r^2 = -0.08$, estimation sample, $n = 147$, $P > 0.05$). A closer examination of the effects of woodland area indicates that the number of bird species was greatest in intermediate-sized woodland patches (Figure 2.2 - All sites). Woodland area had a significant effect on the number of bird species recorded on the transects (Kruskal Wallis: $H = 10.81$, d.f. = 4, $p < 0.05$, Figure 2.2 - All Sites). However, multiple comparison of mean ranks showed no significant area effects between any of the five different size categories ($z = 2.58$, $n = 5$, $0.05 < p < 0.1$). When these same data were re-analysed, dividing the woodland patches into six size categories, the number of species was significantly higher in the intermediate-sized patches (see Chapter 4, Figure 4.1 - All Species).

There were no significant area effects when stringybark woodland and gum-box woodland were analysed separately (stringybark, $F = 2.1$, d.f. = 4, 109, $p = 0.08$; gum-box, $F = 1.2$, d.f. = 4, 92, $p = 0.33$). However, the pattern for both woodland types was similar to that described above (Figure 2.3). This indicates that the greater species richness in intermediate-sized woodland patches was not caused by an interaction between area and woodland type. When the cumulative number of species is considered for the eight seasonal surveys, there were no significant area effects (Kruskal Wallis: $H = 5.5$, d.f. = 4, $p = 0.2$). The regression between the cumulative number of species was also not significant ($r^2 = 0.14$, $n = 63$). There was, however, a tendency towards there being more species in larger woodland patches (Figure 2.4).

If sites with noisy miners were excluded from the analysis, the number of species was significantly higher in woodland patches that were 6 to 20 ha in area, compared with those greater than 100 ha (Figure 2.2 - non-miner sites; $H = 21.96$, d.f. $= 4$, $P < 0.01$; Dunn's multiple comparison technique; $z = 2.81$, $n = 5$, $p < 0.05$). The area of the woodland patch had a significant effect on the number of bird species on transects in noisy miner sites ($H = 12.1$, d.f. $= 4$, $p < 0.05$, Figure 2.2 - Miner Sites). However, although the trend was towards there being fewer bird species in miner territories in patches of woodland that were smaller than 20 ha, *post hoc* tests indicated no significant differences (Dunn's multiple comparison technique; $z = 2.33$, $n = 5$, $0.05 < p < 0.2$). There was a positive correlation between the number of species and the area of the woodland patch in sites where noisy miners were present ($r^2 = 0.41$, $n = 49$, $p < 0.01$, Figure 2.2 - Miner Sites). The opposite trend occurred in sites where noisy miner were absent ($r^2 = 0.29$, $n = 245$, $p < 0.01$, Figure 2.2 - Non-miner Sites).

The number of species in strips of woodland was not significantly different from the number of species in patches that were smaller than 6 ha (Mann-Whitney; $U = 297.5$, $n = 25$ strips and 21 patches, $p = 0.6$, Figure 2.2). Of the 89 bird species that were recorded during the single survey on the 294 sites, 55 (56%) were also recorded on the 26 transects in strips of woodland.

Number of birds

The number of birds increased in sites where large trees and mistletoes were present (Table 2.6). The increased abundance of cut stumps in these sites is an indication that selective logging is taking place where the trees are larger. It is also an indication that there will be more logs present. Cut stumps were more common in smaller patches and closer to human dwellings (Tables 2.3 and 2.4). Sites with more birds also tended to have extensive, tall herbaceous vegetation. A well developed herbaceous vegetation was associated with small isolated patches of gum-box woodland, where there were many mistletoes (Table 2.4). As with bird species richness, the number of birds decreased at sites in large woodland patches, away from water-courses and on sloping country (Table 2.6). The number of birds was also lower in sites affected by dieback and lower where there was greater vegetation species richness. Sites with greater vegetation species richness tended to be in larger woodland patches at lower altitudes. There was greater vegetation biomass in these sites, and subsequently the herbaceous vegetation was reduced (Tables 2.3 and 2.4). Generally the number of birds was greatest in small patches at lower altitude (Table 2.4). The R^2 (ADJ) values from the predictive models of bird abundance ranged from 0.17 to 0.19 (Appendices 2 and 3).

Uncommon Species

This diverse group of 39 species includes raptors with large home ranges such as the brown goshawk and little eagle, species only found in larger patches of woodland such as the satin flycatcher and red-browed treecreeper, ground-foragers such as the peaceful dove and hooded robin, hollow-nesters such as the lorikeets, and seed-eaters such as the finches (Table 2.1). It is not surprising that these species are uncommon on the Armidale Plateau. They were associated with the gum-box woodland at lower altitudes (Table 2.6 - primarily box trees) which has been most extensively cleared by pastoralists. The well developed herbaceous vegetation in gum-box woodland sites (Tables 2.3 and 2.4) probably explains the strong presence of ground-feeders and seed-eaters in this bird group. Uncommon species were associated with a greater percentage of *Acacia* spp. understorey, indicating a sensitivity to grazing. Uncommon species were also negatively affected by dieback, which is more prevalent in the gum-box woodland (Morgan 1981; Lowman and Heatwole 1992). In the present study, however, dieback was greater in stringybark woodland than in the gum-box woodland (Table 2.3). This is because the gum-box woodland existed either as relatively large patches, or very small severely degraded patches. As has been mentioned, most of the severely degraded patches were too small to fit a 200 m strip transect and so are under-represented in this study. Generally, uncommon species were more strongly associated with large woodland patches at lower altitude (Table 2.4). The R^2 (ADJ) values from the predictive models for uncommon species ranged from 0.05 to 0.21 (Appendices 2 to 4).

Common Species

This group of 10 species includes hollow-nesters such as the striated pardalote, fruit-eaters such as the crimson rosella, open-country species such as the eastern rosella, summer visitors and foliage-gleaners such as the rufous whistler, understorey species such as the grey fantail, bark-foragers such as the white-throated treecreeper, and territorial species such as the fuscous honeyeater (Table 2.1). The pattern of distribution of common bird species was similar to that for bird species richness and number of birds (Table 2.4). Common species were positively associated with stringybark woodland at higher altitudes, where large trees, mistletoes, cut stumps, and logs were present (Table 2.6). They were also more common where there was greater tree density and increased canopy cover. Common species were less abundant at sites in larger woodland patches that tended to be away from permanent water and farm buildings, and away from the edge of the woodland patch. Common species were also negatively associated with dieback and distance to water-courses. The R^2 (ADJ) values for the models predicting the abundance of common species varied from 0.16 to 0.18 (Appendices 2 to 4).

Species With a Low Population Density

This group of 11 species includes forest raptors such as the brown goshawk, hollow-nesters such as the dollarbird, ground-foragers such as the hooded robin and speckled warbler, bark-foragers such as the crested shrike-tit and grey shrike-thrush, summer visitors, and foliage-gleaners such as the rufous whistler and white-throated warbler (Table 2.1). Species with low population density were more likely to occur in stringybark woodland where logs were present, and where there was extensive herb cover (Table 2.6). There were, however, some apparent inconsistencies between the selected predictors for this bird group. Species with low population density were less common in sites at higher altitudes that were generally associated with stringybark woodland. Furthermore, these species were less common in sites away from farm buildings where logs were more common. Another inconsistency is that extensive herb cover is associated with gum-box rather than stringybark woodland (Tables 2.3 and 2.4). These inconsistencies suggest that species with lower population density are perhaps being forced into marginal habitat. It is possible that these species are ideally suited to a habitat that no longer exists on the Armidale Plateau. That is, gum-box woodland on the richer soils, with an extensive herbaceous vegetation and numerous logs. For use as firewood and fence-posts, the timber from gum and box trees is preferred to that from stringybark trees (Wall pers. comm.). It is possible that the removal of logs from gum-box woodland sites may have forced these species to exist in stringybark woodland where logs still exist (Tables 2.3 and 2.4). It is also possible that territorial species with a high population density, such as the noisy miner and fuscous honeyeater, drive these species out of the gum-box woodland into the marginal stringybark woodland (see below, also Chapter 3). Species with a lower population density appear to be tolerant to low levels of logging. They were less common in sites affected by dieback and less common on the sloping country, which is likely to be nutrient-poor and has less herbaceous vegetation (Table 2.4). Under the present circumstances on the Armidale Plateau, it appears that species with low population density are more common in larger woodland patches at lower altitudes (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of species with low population density varied from 0.06 to 0.16 (Appendices 3 and 4).

Species with a High Population Density

This group of 11 species includes hollow-nesters such as the musk lorikeet, open-country species such as the eastern rosella and common starling, foliage-gleaners such as the striated thornbill, territorial species such as the noisy miner and fuscous honeyeater, understorey species such as the superb fairy-wren, and ground-foragers such as the white-winged chough. A number of these species are also flocking species and co-operative breeders (Table 2.1). Species with a higher population density were more common at sites in gum-box woodland at lower altitude, where the herb vegetation was extensive and tall (Table 2.6). They also tended to be more common in sites away from farm buildings and where dead trees had not been removed. Species with higher population densities were less common in larger patches away from the edge of the patch, where there was a greater number of vegetation species and where more logs were present (Table 2.6). There is an inconsistency with sites in larger woodland patches also being further from farm buildings (Table 2.3). This indicates that distance to buildings is not a reliable predictor for this bird group. The general trend, however, is that species with a high population density were more common in smaller woodland patches at lower altitude, that tended to be closer to farm buildings (Table 2.4). Species with a high population density were also less common where logs were present (HMDS only). The R^2 (ADJ) values for the models predicting the abundance of species with high population density varied from 0.07 to 0.36 (Appendices 2 to 4).

Woodland Species

These 74 remnant dependent species (Table 2.1) were more common in sites where mistletoes were present and where a greater proportion of *Acacia* spp. understorey was present (Table 2.6). These species showed a similar pattern to that shown by bird species richness, except that being woodland-dependent, this bird group was more likely to be affected by patch isolation and so more likely to occur in larger woodland patches (Table 2.4). The HMDS indicates that woodland species were associated with stringybark woodland at higher altitudes, where logs were present and where the vegetation biomass was greater. This analysis also indicates that woodland species were less common in sites that were further from the edge of the patch (Table 2.6). There is a contradiction with the standard multiple regression using a single spring survey (SMR - Spring), indicating that woodland species were more common at lower altitudes. This result goes against the trend towards greater bird species richness in stringybark woodland at higher altitude, and indicates that altitude is not a reliable predictor of woodland-dependent bird species richness. The R^2 (ADJ) values for the models predicting the abundance of woodland species varied from 0.08 to 0.28 (Appendices 2 to 4).

Open-country Species

Open-country species include ground-foragers such as the eastern rosella, galah, red-rumped parrot and common starling, which also nest in hollows, and ground-foragers such as the rufous song-lark and yellow-rumped thornbill and Australian magpie lark which do not use hollows (Table 2.1). Open-country species were more common in sites where the herbaceous vegetation was extensive and tall, and where the trees were affected by dieback (Table 2.6). The HMDS indicates that open-country species were also more common at higher altitudes where cut stumps were present. The reduction of canopy cover as a result of dieback and selective logging was more likely to occur in smaller woodland patches at higher altitudes (Tables 2.3 and 2.4). This is likely to result in more vigorous herb growth which favours ground-foraging open-country species. This result suggests that open-country species should have been more common in the stringybark woodland, which occurs at higher altitude, has more cut stumps present and was more likely to be affected by dieback. However, the analyses also indicate that open-country species were less common on the sloping country where stringybark woodland was more likely to occur (Table 2.6). It appears that while open-country species are favoured by dieback and selective logging, they will still avoid the poorest soils on the sloping country, where only stringybark woodland can grow. Open-country species were less common in large patches of woodland, away from the edge of the patch, where there was greater vegetation species richness, greater tree density and increased canopy cover (Table 2.6). In summary, open-country species were most common in smaller woodland patches at lower altitude (Table 2.4), particularly in sites where the pasture has been improved and grazing pressure is not too heavy. The R^2

(ADJ) values for the models predicting the abundance of open-country species varied from 0.18 to 0.42 (Appendices 2 to 4).

Ground-foragers

This group of 16 species includes species that appear to be dependent on extensively wooded areas, such as the peaceful dove and common bronzewing, hollow-nesters such as the sacred kingfisher and laughing kookaburra, as well as the robins and finches (open-country species excluded, Table 2.1). The HMDS indicates that these species were more common where there was greater tree density, increased herb height and where box mistletoes were present. This same analysis indicates that ground-foragers were less common at higher altitudes, where there is increased shrub cover and where dieback was present (Table 2.6). The negative association between ground-foragers and shrub cover is due to the fact that an increased presence of shrubs results in a reduction of herb vegetation (Tables 2.3 and 2.4). The increased abundance of ground-foragers where there was greater tree density was, however, inconsistent with this result. It may be that the greater vegetation biomass in sites where there was a greater tree density, provided protection for many of the smaller species included in this group of ground-foragers. The negative association between ground-foragers and altitude is probably due to a preference for the richer soils at lower altitude. This suggests that ground foragers would be more common in gum-box woodland, a conclusion supported by the positive association between this group and box mistletoes, the common mistletoe on gum and box trees. There is, however, an inconsistency with this group also being negatively associated with the proportion of gum trees. This indicates that woodland type is not a good predictor of the abundance of ground-foragers. In general, these species appear to be more common in small patches at lower altitudes (Table 2.4). The R^2 (ADJ) value for the regression analysis using seasonal surveys was 0.24 (Appendix 4).

Understorey Species

This group of seven species includes honeyeater species such as the yellow-faced honeyeater, ground-foragers such as the superb fairy-wren, and seed-eaters such as the finches (Table 2.1). Understorey-dependent species were more common where there was greater vegetation biomass and where a diverse mix of shrub and sub-canopy species were present. These species were also more common in stringybark woodland at higher altitudes where the drooping mistletoe was present (Table 2.6). The presence of a greater proportion of *Acacia* spp. understorey, indicates that grazing by stock is less intense where understorey-dependent species were more common. The HMDS indicates that understorey-dependent species tend to occur in larger patches of woodland, away from farm buildings and permanent water such as farm dams. These species were less common where dieback and where box trees were present. The increased abundance of understorey-dependent species at higher altitudes may be due to honeyeaters, such as the yellow-faced honeyeater, responding to the flowering of the drooping mistletoe in the stringybark woodland during spring. It may also be due to the increased presence of large trees and logs, and the more extensive herbaceous vegetation cover in the larger woodland patches at higher altitudes (Tables 2.3 and 2.4). However, vegetation species richness and (although not significantly) the percent shrub cover actually decreased at higher altitude. In summary, understorey species, like the woodland-dependent species, were more common in larger patches at higher altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of understorey-dependent species varied from 0.08 to 0.46 (Appendices 2 to 4).

Bark-foragers

This group of seven species includes hollow-nesters such as the treecreepers, species that also feed on the ground such as the brown treecreeper and grey shrike-thrush, and the carbohydrate-feeding white-eared honeyeater (Table 2.1). These species were more common in sites with large, tall trees where logs were present, and where there was a diversity of vegetation species (Table 2.6). Bark foragers were also more common where *Acacia* spp. shrub vegetation was more extensive, indicating an association with sites in which grazing pressure was reduced. The HMDS also indicates that these species were more common at higher altitudes, where there was greater tree density, and where mistletoes were present. Bark foragers were less common where dieback was present, and showed neither a preference for stringybark woodland nor gum-box woodland. Some species in this group, such as the red-browed treecreeper, are more common where gum trees occur, whereas others such as the white-throated treecreeper are associated with stringybark woodland (Noske 1985). This suggests that bark-foragers are most common where there is an equal mix of gum-box woodland and stringybark woodland, which was more likely to occur in large woodland patches at lower altitudes (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of bark-foragers species varied from 0.03 to 0.29 (Appendices 2 to 4).

Foliage-gleaners (and Snatchers)

This group of 13 species includes small domed-nesting insectivores such as the thornbills, the striated pardalote, a hollow-nester, and the spotted pardalote a ground-nester, as well as the cuckoos and whistlers. The honeyeaters were excluded from this group (Table 2.1). Foliage-gleaners were more common in stringybark woodland where mistletoes and logs were present. They were also more common where tree density was greater and where there was increased vegetation biomass (Table 2.6). They were less common at lower altitudes, in larger patches of gum-box woodland, away from the woodland edge and away from human settlement. Foliage-gleaners were also less common where there was greater vegetation species richness and where trees were affected by dieback. Generally, as with bird species richness, foliage-gleaners were more common in smaller woodland patches at higher altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of understorey-dependent species varied from 0.05 to 0.15 (Appendices 2 to 4).

Fruit-eaters

This group of six species includes the mistletoebird, and hollow-nesters, such as the king parrot and crimson rosella. Open-country species, as well as the pied currawong, an omnivorous species which includes fruit in its diet, were not included in this group (Table 2.1). Fruit-eaters were more common in larger woodland patches, away from the edge of the woodland patch and human settlement (Table 2.6). They were also more common where tree density was greater, where the canopy cover was extensive, and the trees were large. Sites with fruit-eaters present tended to have greater vegetation biomass, logs and mistletoes present. Fruit-eaters were less common where dieback was present. In summary, fruit-eaters were associated with large woodland patches at higher altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of fruit-eaters varied from 0.06 to 0.19 (Appendices 2 to 4).

Honeyeaters

This group of 13 species includes understorey-dependent species such as the yellow-faced honeyeater, summer visitors such as the noisy friarbird, and winter visitors such as the yellow-faced honeyeater. Noisy miners were excluded from this group (Table 2.1). Honeyeaters were more common in larger woodland patches, away from the edge of the patch, and away from human settlement (Table 2.6). They were also more common at sites where there were more large trees, more dead trees, greater tree density, greater vegetation biomass and greater vegetation species richness. Collectively, these environmental features are associated with larger woodland patches (Tables 2.3 and 2.4). Honeyeaters were also more common in sites where mistletoes were present, which tended to be smaller isolated woodland patches.

There were some inconsistencies between the different analyses. The stepwise regression using principal components (Table 2.6 - PCA) indicates that honeyeaters were more common where dieback was present, yet the standard multiple regression using seasonal surveys (SMR - Seasonal) indicates that honeyeaters were less common in sites affected by dieback. It may be that honeyeaters tolerate certain levels of dieback in smaller patches of woodland because there are more mistletoes in these sites. There was a second inconsistency, with the HMDS indicating that honeyeaters were more common at higher altitudes in the stringybark woodland where there were more drooping mistletoes present. The standard multiple regression using a single spring survey, also indicates that honeyeaters were associated with the drooping mistletoe which occurs in stringybark woodland. However, the standard multiple regression using a single spring survey (SMR - Spring) and the regression analysis using principal components, both indicate that honeyeaters were less common in stringybark woodland. Woodland type is likely to be a seasonal predictor of honeyeater abundance. It appears that honeyeaters were more common in the stringybark woodland during winter and spring, yet more common in the gum-box woodland during the summer and autumn. This is likely to be a response to the alternative flowering of the drooping mistletoe and the box mistletoe (Spearman rank correlation between the proportion of stringybark woodland and the number of drooping mistletoes was $r = 0.43$, and between the proportion of gum-box woodland and number of box mistletoe was $r = 0.34$, $p < 0.05$, see also Chapter 3). It is noteworthy that woodland type was not a predictor in the standard multiple regression model based on repeated seasonal surveys (Table 2.5, SMR - Seasonal). This was probably due to the alternate seasonal changes in abundance cancelling each other out.

Honeyeaters were also more common on sloping country closer to water-courses (Table 2.6). Honeyeaters as a group are well represented by winter visitors, and as a group are relatively mobile on a local and regional scale. It may be that as well as being attracted to the nutrient-rich habitat near water-courses, honeyeater species are using the water-courses as fly-ways (see also Chapter 3). Generally, honeyeaters were more common in larger woodland patches at higher altitudes (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of honeyeaters varied from 0.07 to 0.57 (Appendices 2 to 4).

Nest Predators

This group of seven species includes open-country species such as the Australian raven, hollow-nesters such as the laughing kookaburra, ground-foragers such as the grey shrike-thrush and Australian magpie, and omnivorous species such as the pied currawong (Table 2.1). Nest predators were less common in large patches of woodland, in sites that were further from the edge of the patch, and further from farm buildings and farm dams (Table 2.6). The HMDS indicates that they were also less common in sites where there were more vegetation species and less common at lower altitudes. Vegetation species richness was greatest in the gum-box woodland at lower altitudes, indicating that nest predators were more common in the stringybark woodland. In summary, nest predators were more common in small patches of woodland at higher altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of nest predators varied from 0.01 to 0.14 (Appendices 2 and 4).

Hollow-nesters

This group of 13 species includes the striated pardalote, a foliage-gleaner, ground-foragers such as the sacred kingfisher and laughing kookaburra, and bark-foragers such as the treecreepers. Open-country species were excluded from this group (Table 2.1). Hollow-nesters were more common in gum-box woodland where there were numerous large trees and a dense canopy cover (Table 2.6). These sites tended to have an extensive, tall herbaceous vegetation and a diversity of vegetation species present (Tables 2.3 and 2.4). The increased abundance of logs and increased proportion of *Acacia* spp. understorey, indicate that these sites were less heavily grazed. Although area was not selected as a predictor, hollow-nesters were more likely to be in large woodland patches because of their tendency to be in sites further from farm buildings. It is likely that this aversion to human settlement is a result of the increased logging activity in sites close to human settlement (Tables 2.3 and 2.4). Hollow-nesters were less common where dieback was present. In summary, hollow-nesters were more common in large woodland patches at lower altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of hollow-nesters varied from 0.10 to 0.18 (Appendices 2 to 4).

Summer Visitors

This diverse group of 22 species includes ground-foragers such as the rufous song-lark, hollow-nesters such as the dollarbird, foliage-gleaners such as the rufous whistler, understorey species such as the grey fantail, fruit-eaters such as the mistletoebird, and honeyeaters such as the noisy friarbird (Table 2.1). Summer visitors were more common where there are large trees, many mistletoes and an extensive, tall herbaceous vegetation. This suggests that summer visitors may be more common in smaller woodland patches. However, the HMDS indicates that this bird group was also more common where logs were present and where there was greater vegetation species richness (Table 2.6), environmental features which tend to be associated with larger woodland patches (Tables 2.3 and 2.4). The regression analysis using the principal components also indicates that summer visitors were more common in stringybark woodland. However, it appears that woodland type is not a reliable predictor. Although the HMDS indicates that summer visitors are less common in gum-box woodland, this same analysis suggests that these species are more common where there are box mistletoes. The HMDS also indicates that summer visitors are less common at higher altitudes where the sites tended to be in stringybark woodland (Table 2.3). Summer visitors were less common where dieback was present and less common at sites further from roads. It is possible that summer visitors are attracted to the well developed herbaceous vegetation in the nutrient-rich gum-box woodland habitat, that tends to occur in roadside vegetation (Table 2.3). It may also be that summer visitors are using the roadside vegetation, which tends to be linear, as fly-ways for movement (Chapter 3). It is most likely that summer visitors were associated with small woodland patches at lower altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of summer visitors varied from 0.10 to 0.22 (Appendices 2 to 4).

Winter Visitors

This group of six species is strongly represented by the honeyeaters, and also includes foliage-gleaners such as the golden whistler and spotted pardalote (Table 2.1). Winter visitors were more common in stringybark woodland at higher altitude, where mistletoes were present, where there was a mixture of vegetation species (in the understorey only) and greater vegetation biomass (Table 2.6). These species were also more common in larger patches of woodland and where logs were present. Winter visitors were less common at sites away from water-courses. It may be that winter visitors are attracted to the moister microhabitat that is associated with water-courses (winter is the low rainfall period on the Armidale Plateau), or the reduced logging pressure in these sites (Tables 2.3 and 2.4). It may also be that winter visitors use water-courses as fly-ways (see Chapter 3). In summary, winter visitors were more common in larger woodland patches at higher altitude (Table 2.4). The R^2 (ADJ) values for the models predicting the abundance of summer visitors varied from 0.08 to 0.42 (Appendices 2 to 4).

Resident Species

Resident species were more common in sites with a greater percentage of *Acacia* spp. understorey, indicating a sensitivity to prolonged grazing pressure by stock. The HMDS indicates that these species were also more common in gum-box woodland where logs and box mistletoe were present (Table 2.6). Residents were less common at higher altitudes in sites affected by dieback. These site characteristics suggest that resident species are more common in larger patches at lower altitudes (Tables 2.3 and 2.4). The R^2 (ADJ) values for the models predicting the abundance of summer visitors varied from 0.04 to 0.24 (Appendices 2 and 4).

2.6 DISCUSSION

This study has demonstrated that a range of environmental variables is likely to be responsible for structuring bird assemblages on the Armidale Plateau. Furthermore, the interaction between these environmental variables is likely to be as important as their individual effects. Although the area of the woodland patch was not the most important predictor, it is useful for the purposes of this discussion to consider bird assemblages in large compared with small woodland patches. Because of the historical differences between the management of the richer more productive soils at lower altitude, and the rocky nutrient-poor soils on the ridge-tops, it is also useful to consider bird assemblages in relation to altitude. It should be noted, however, that while particular environmental variables are associated with either large or small woodland remnants, or with high or low altitudes, they are not exclusive to either. A summary of the conclusions from this chapter is placed at the end of this discussion.

2.6.1 Area and Bird Species Richness

The predictive models based on the single survey in 294 sites indicate that bird species richness, as well as the abundance of a number of other bird groups, was negatively correlated with area. Furthermore, bird species richness in sites that were surveyed once, was greatest in the intermediate-sized woodland patches, decreasing in areas that were larger than 400 ha or smaller than 6 ha. The repeated surveys were more likely to include the rarer, less evenly distributed species, that are more common in the larger woodland patches (Chapter 4). The single bird survey on the other hand, is likely to favour the more common and conspicuous species that occur in smaller woodland patches.

The single survey also indicates that most bird groups on the Armidale Plateau were not affected by patch isolation (with the exception of woodland-dependent species). Howe (1984) also found no isolation effects on bird assemblages in forest fragments on the

New England Tablelands. It is likely that a large portion of the bird species on the Armidale Plateau utilise the area between the woodland remnants as habitat (albeit poor quality habitat), and for dispersal. However, this does not mean that bird species will move freely across large areas of open open-country on the Armidale Plateau. Rather, they appear to use strips of vegetation as fly-ways (Chapter 3), and probably 'tree-hop' using isolated trees between woodland remnants (Date *et al.* 1991).

While the importance of large areas of habitat for species conservation is well documented (see Chapter 1 references), the role of smaller remnants in conserving species is less well understood. On the Armidale Plateau it appears that during any given season, most of the bird species richness is in woodland patches as small as 20 ha. The seasonal surveys indicate that these smaller remnants are most important as reservoirs for species richness during spring and autumn, when many birds are moving through the landscape (Chapter 3). Other studies have found that birds readily occur in small habitat remnants throughout disturbed landscapes (Howe *et al.* 1981; Loyn 1985a, 1987; Grover and Slater 1994). These studies, as well as the present study, suggest that it should be possible to conserve a large proportion of the avifauna in agricultural landscapes where the option of acquiring further large areas of habitat for conservation is limited.

The importance of smaller woodland remnants for maintaining species richness on a seasonal basis on the Armidale Plateau, is likely to be dependent on the continued existence of the more extensively wooded areas. In a review of the subject, Andrén (1994) concluded that the original habitat must be cleared beyond a critical point (about 70% of the original habitat) before the loss of species as a result of area and isolation effects will be greater than that due to habitat loss alone (see also Green 1994). Although the present woodland cover on the Armidale Plateau is between 20 and 30%, the woodland and forest cover over the whole of the New England Tablelands is about 50%. This suggests that the relatively minor response of birds to area and isolation, best illustrated by the single survey in the 294 sites, may only be sustained to a critical point. As tree loss continues beyond this critical point, large patches of woodland that are not isolated will become increasingly important for the conservation of bird species diversity on the Armidale Plateau Also, the effects of fragmentation on the avifauna will increase regionally.

With further clearing of woodland and increasingly intensive land-use, the following environmental features appear to have an escalating impact on bird assemblages on the Armidale Plateau. The impact happens to a degree in larger woodland patches but is most marked in the smallest woodland patches, particularly those less than six hectares.

2.6.2 Dieback: Vegetation Biomass

Dieback is most vigorous along the edges of woodland remnants because the majority of leaf-eating insects come from the surrounding pasture (Lowman and Heatwole 1992). As such, dieback is worse in the smaller isolated woodland patches where there is more edge habitat (Table 2.3; see also Ford and Bell 1981; Morgan 1981). Most of the shrub and sub-canopy cover on the Armidale Plateau was regenerating eucalypts, which are particularly susceptible to dieback. This partly explains the negative association between dieback and both percent shrub cover and vegetation biomass (Table 2.3).

Four bird groups were positively associated with larger woodland patches - understorey-dependent species, fruit-eaters, honeyeaters and winter visitors. A feature that is common to these four groups is that they were all more common in sites where there was increased vegetation biomass (Tables 2.4 and 2.6). This suggests that one of the primary effects of reducing the area of woodland patches on the Armidale Plateau is the subsequent reduction of vegetation biomass. This supports the conclusion by Ford and Bell (1981) that the reduced bird species richness in the smallest woodland remnants on the Armidale Plateau is partly a result of reduced foraging opportunities, where the vegetation biomass has been removed by dieback. In southeast Queensland, Grover and

Slater (1994) found that understorey species and raptors were more common in larger remnants. In rainforest on Barro Colorado Island, Panama, Karr (1982) also found that understorey-dependent species were particularly susceptible to habitat fragmentation. Fruit-eaters have been found to be particularly sensitive to habitat disturbance by Hutto (1989) and Kattan (1991). While this result is consistent with the present study, Newmark (1990) found that fruit-eaters were more common than expected in smaller fragments of tropical forest in Tanzania.

2.6.3 Grazing and Vegetation Biomass

It is also likely that shrub vegetation is removed by stock grazing, which is more intensive in the smaller woodland patches (Ford and Bell 1981; Morgan 1981, Ford *et al.* 1995). Shrub vegetation on the Armidale Plateau provides food, cover and nesting habitat for many bird species (Dunkerley 1989; Davidson and Davidson 1992). Despite this, understorey alone was a relatively unimportant predictor of bird abundance. This may be because of the low level of understorey cover throughout the Armidale Plateau (average cover less than 10%, Table 2.2). In southeast Queensland, Grover and Slater (1994) found that the number of forest bird species and understorey species only increased when the percent cover of understorey vegetation increased beyond a critical value of about 60%. The presence of shrub cover on the Armidale Plateau is likely to play a role in structuring bird species assemblages, when in combination with other variables such as dieback, grazing pressure, and the area of woodland patch. In northeastern New South Wales, Howe *et al.* (1981) found that grazing in smaller remnants of subtropical rainforest and the resulting removal of understorey vegetation reduced bird species richness. They noted that understorey-dependent species or species foraging close to the ground were particularly susceptible. The pattern of increased bird species richness or greater number of birds where there is greater vegetation biomass, seems to be true for other areas in New South Wales (Milledge and Recher 1985), Queensland (Leach and Recher 1993), Victoria (Loyn and Suckling 1984; Loyn 1985b, 1987) and in the Western Australian wheatbelt (Cale 1990).

2.6.4 Grazing and Herbaceous Vegetation

In most cases it can be expected that prolonged, heavy grazing leads to a loss of bird species (Recher and Lim 1990; Bock *et al.* 1992; Garnett 1992; Ford 1993). However, where grazing is absent the herbaceous vegetation is reduced due to the unchecked growth of the understorey vegetation (Table 2.3). Moderate levels of grazing increase the amount of herb cover without removing all of the understorey vegetation. The positive association between tall, extensive herb vegetation and ground-foragers and summer visitors, as well as bird species richness, suggests that moderate levels of grazing are preferable to no grazing at all for some bird groups. It follows that the loss of bird species on the Armidale Plateau is not caused by grazing *per se*, but rather that grazing is too heavy and too widespread. Grazing is also too persistent on the Armidale Plateau, giving pastures little chance to recover. This situation, where bird species may or may not be favoured by grazing depending on the grazing intensity, is similar to that described by Saunders and Curry (1990) for the Murchison River pastoral region in Western Australia. This result supports the suggestion that as long as all vegetation layers are present, an extensive herb layer leads to increased bird species richness, simply because it results in a more complex habitat (Willson 1974; Recher 1993). However, the present study contradicts a previous study on the Armidale Plateau in which bird species richness was found to decrease where the herb cover was more extensive (Howe 1984). This decrease, however, only occurred in the continually forested areas (control plots), and as previously mentioned, was based on repeated surveys in open forest rather than woodland.

The well developed herb vegetation in smaller woodland patches explains the increased abundance of open-country species including ground-foragers, in areas affected by dieback (Table 2.6). While this may be true, it should be noted that ground-foragers that were not open-country species, such as the peaceful dove, sacred kingfisher as well as the robins and finches, were less common on sites affected by dieback. An increased abundance of open-country species in smaller woodland remnants was described in previous studies on the New England Tablelands (Howe 1986; Howe *et al.* 1981), as well as in *Melaleuca* remnants near Brisbane (Grover and Slater 1994), and in Victoria (Loyn 1985a).

2.6.5 Vegetation Species Richness

As well as maintaining a mixture of herb and shrub vegetation, moderate levels of grazing maintains vegetation species diversity in both these vegetation layers (Duffy 1974; Miller and Watson 1974; Connell 1979). The number of vegetation species was closely associated with the vegetation biomass, as well as the proportion of *Acacia* spp. in the shrub vegetation. These environmental features were all greater in larger woodland patches (Tables 2.3 and 2.4). Habitats where vegetation species richness was greater are generally associated with greater bird species richness (James 1971; Gilmore 1985; Mac Nally 1990b; Leach and Recher 1993). A previous study on the New England Tablelands by Howe (1984) also found there to be a positive relationship between bird species richness and the number of vegetation species. However, in the present study, bird species richness was not positively associated with vegetation species richness. On the Armidale Plateau, vegetation species diversity, most of which was in the shrub layer, was greatest in the gum-box woodland (Tables 2.3 and 2.4). As is discussed below, the bird species richness was lower than expected in this woodland type. While bird species richness was not significantly correlated with vegetation species richness, bird groups that were more common in larger woodland patches; such as the honeyeaters, understorey-dependent species, winter visitors, bark-foragers and hollow-nesters, were all positively associated with vegetation species richness (Tables 2.4 and 2.6). The increased vegetation biomass that is associated with greater vegetation species diversity, results in less herbaceous vegetation. This explains why fewer open-country species are found in sites where there is greater vegetation species richness.

Stock graze the nitrogen-rich *Acacia* spp. preferentially, and where grazing has been excessive these understorey shrubs are less common (Scougall *et al.* 1993). The percentage of *Acacia* spp. understorey was the only predictor (+ ve) for uncommon species, based on the repeated surveys (Table 2.6). The positive association between this group of birds that were least common on the Armidale Plateau, and the proportion of *Acacia* spp. understorey, further supports the conclusion that over-grazing is a reason for the decline of bird species on the Armidale Plateau.

2.6.6 Timber Removal

The present study is in agreement with Davidson and Davidson (1992) who emphasise the importance of fallen timber to wildlife in pastoral areas on the New England Tablelands. Bennett (1992) and Recher (1993) have also commented on the importance of retaining logs in roadside vegetation as a means of restoring remnant habitat. The proximity of gum-box woodland to roads (Table 2.3) has resulted in much of the fallen timber being cleared (Wall pers. comm.). It also appears that logs were more likely to be left to break down naturally in stringybark woodland. The greater number of cut stumps in stringybark woodland compared with gum-box woodland (Table 2.3) was partly due to greater tree density in stringybark woodland. It is also due to the smallest, most degraded, gum-box woodland sites being under-represented in this study. The positive association between stringybark woodland and logs, may partly explain the increased bird species richness in this woodland compared with gum-box woodland (see below).

The present study is also in agreement with previous studies which indicate that bird species that are dependent on large trees are particularly susceptible to habitat clearing and fragmentation (Saunders *et al.* 1985, 1987; Robinson 1991; Garnett 1992; Stokes in prep.). Collectively, tree diameter, canopy height and the presence of large trees were as important as predictors of bird abundance (species richness as well as the abundance of particular bird groups), as were the percent dieback, number of mistletoes and altitude

(Table 2.7). In all cases, these three environmental features that are associated with the presence of large trees were positive predictors of bird species richness and bird abundance. The reduced abundance of hollow-nesting species in sites closer to buildings (Tables 2.4 and 2.6, open-country species were excluded from this group) indicates that the large trees that have been left as shade trees for stock are unsuitable as habitat for this bird group. This is supported by the hollow-nesters being positively associated with canopy cover and negatively associated with dieback, indicating that they require healthy trees (Table 2.6). Not only are large trees being cut down in the smaller remnants where the trees are more accessible, but the large trees which are left as shade for stock are degraded by dieback. As a result, it is likely that some hollow-nesters are particularly susceptible to woodland fragmentation on the Armidale Plateau. The increased grazing pressure in these smaller sites also means that there is no tree replacement, a situation that has also been described in Victoria by Bennett *et al.* (1994).

2.6.7 Edge Effects

Edge effects were closely associated with, and partly responsible for area effects (Table 2.5 - PC 7, Table 2.4). The distance to the edge of the woodland patch was a negative predictor of species richness only when the effects of this environmental variable were combined with the effects of area (Table 2.6 - PC7). When considered separately (as in the HMDS and regression analyses), both the area of the patch and the distance to the edge of the patch had similar effects; bird groups that were less common in larger woodland patches tended to be negatively associated with the distance to the edge of the woodland patch. However, the importance of distance to edge of woodland patch as a predictor was greatly reduced (Table 2.6). This lack of edge effect, when considered independently of area, is probably due to the open nature of the woodland habitat (< 30% canopy cover), as well as the diffuse or variegated nature of the landscape on the Armidale plateau. It is also likely that the increased presence of noisy miners along edges and in strips of woodland (unpublished data), partly offsets the increased bird species richness along edges. Other Australian studies have described changes in bird communities associated with edges that are similar to those found in this study (Loyn 1985a; Catterall *et al.* 1991; Lynch and Saunders 1991).

2.6.8 Noisy miners

The increased abundance of noisy miners in smaller woodland patches that are affected by eucalypt dieback has been previously documented by Ford and Bell (1981) on the Armidale Plateau, as well as in Victoria by Loyn *et al.* (1983). It appears that the presence of territorial 'edge species such as the noisy miner, increases the importance of large woodland remnants for maintaining bird species richness. The single bird survey in 294 woodland sites indicated that bird species richness was negatively associated with the area of the woodland patch. However, this pattern was reversed when only sites with noisy miners were considered (Figure 2.2 - compare Miner and Non-miner Sites). In a study of bird communities in the New Guinea islands, Diamond (1979) predicted that 'tramp and super-tramp' species would be more common in smaller habitat islands. He characterised these species as generalists in their habitat requirements, good dispersers but poor competitors. The noisy miner is a generalist species and a good disperser. However, it is also a good competitor in the smallest woodland patches. The present study suggests that the apparent spread of noisy miners on the Armidale Plateau (Chapter 4) may be checked by increasing the minimum woodland patch size throughout the landscape to above 20 ha (Figure 2.2 - Miner Sites). It is also likely that the increased abundance of noisy miners in smaller woodland patches on the Armidale Plateau is partly caused by the removal of the understorey vegetation by dieback and high grazing pressure (Loyn 1987; Dunkerley 1989; Catterall *et al.* 1991; Grey 1995).

It has been suggested that the noisy miner may contribute to the spread of eucalypt dieback in smaller forest patches by excluding other insectivorous birds, resulting in a smaller overall harvest of insects by the bird community (Dow 1977; Ford 1993; Loyn *et al.* 1983). Experiments on the Armidale Plateau in which eucalypt branches, were caged to exclude foraging birds, suggest that bird assemblages outside miner territories remove 55 to 70% of the leaf-eating insects (Ford *et al.* 1985; Dunkerley 1989; Ford 1989). Similar outbreaks of leaf-eating insects, in caged relative to uncaged branches were recorded on the Southern Tablelands near Canberra (Landsberg *et al.* 1990). These experiments suggest that by establishing territories and excluding most other insectivorous birds, noisy miners are effectively 'caging' smaller woodland remnants on the Armidale Plateau, resulting in an increase in abundance of leaf-eating insects and an outbreak of dieback.

2.6.9 Nest Predators

Another possible reason for the decline in bird species richness in the smaller woodland patches on the Armidale Plateau is the increased abundance of nest predators in these sites (Tables 2.4 and 2.6). It may be that nest predators are more common in the smaller patches because of the increased foraging efficiency associated with small habitat patches (Temple and Cary 1988). In addition, the reduced vegetation biomass in the smaller woodland patches, as a result of dieback and grazing, is likely to make it easier for predators to search for nests (Askins and Philbrick 1987; Martin and Roper 1988; Santos and Telleria 1991). It may also be that nest predators were more common in the smaller woodland patches because smaller patches were closer to farm buildings and permanent water (Saunders and Curry 1990). Human settlements provide alternative food sources for nest predators. For example, introduced berry-producing shrubs that are often planted near farm houses on the Armidale Plateau are likely to have resulted in an increased abundance of pied currawongs (Bass 1990; Ford 1993). Garbage dumps have also been found to cause an increased abundance of avian and mammalian nest predators (Angelstam 1986; Terborgh 1989).

Numerous studies have described an increase in the abundance of potential nest predators in smaller habitat remnants; in Australia (Howe 1986; Loyn 1987; Catterall *et al.* 1991), as well as in the Northern Hemisphere (Ambuel and Temple 1983; Andrén *et al.* 1985; Martin 1988; Small and Hunter 1988; Møller 1988, 1989). It has been suggested that the increased presence of nest predators in disturbed, fragmented landscapes is a likely explanation for the decline of many small songbirds in rural landscapes (Robbins 1980; Wilcove 1985; Terborgh 1989). To some extent this appears to be happening on the Armidale Plateau (see also Ford and Bell 1981; Ford 1993). A three year survey of bird nests on the Armidale Plateau, indicated that few of the smaller passerines nested in strips of woodland and woodland remnants smaller than 6 ha; those that did were usually unsuccessful (Barrett *et al.* in prep.).

2.6.10 Mistletoes

It is likely that eucalypts will be killed by mistletoes in sites where heavy infestations occur. It may be that mistletoes flourish where tree health is low (Reid pers. comm.). This view is supported by the increased abundance of mistletoes where the canopy cover was reduced and where there were fewer dead trees (Table 2.3). Mistletoes are tolerant to habitat disturbance up to a point, being positively associated with smaller woodland patches, woodland edges, farm buildings, reduced canopy cover and cut stumps. As a result, it is likely that the negative effects of human impact may be initially offset by the increased abundance of mistletoes in these disturbed sites. However, unless mistletoe abundance is controlled by fire or grazing (possums eat mistletoe foliage, Reid pers. comm.), they are likely to have a detrimental effect in the longer term. The presence of mistletoes was such a strong predictor, that at times it overrode most other environmental variables including dieback (Table 2.6 - honeyeaters). Both summer and winter visitors were positively associated with mistletoes, suggesting that they provide a seasonal food source for these two bird groups. It appears that honeyeater species move into and out of woodland patches, and possibly between the gum-box and stringybark woodland, in response to the seasonal flowering of mistletoes (see also Chapter 3 and Ford *et al.* 1990).

2.6.11 Altitude and Woodland Type

The response of the bird community to altitude is likely to depend on species preferences for either stringybark woodland or gum-box woodland (Tables 2.4 and 2.6). There were, however, fewer patches of gum-box woodland than stringybark woodland, because gum-box woodland occurs on the richer soils at lower altitude, and has been extensively cleared for pasture. The general associations between particular bird groups and woodland type identified in the present study, were consistent with a previous study of the foraging behaviour of individual bird species on the Armidale Plateau (Ford *et al.* 1986). The positive association between the gum-box woodland at lower altitudes and uncommon bird species (Table 2.5), highlights the importance of protecting areas of woodland that occur on the more productive soils at lower altitude. Ground-foraging birds as a group were also associated with smaller woodland patches in the more productive habitats at lower altitudes (Tables 2.4 and 2.6). However, these species were also sensitive to the disturbances associated with small patches of gum-box woodland at lower altitude, such as dieback and heavy grazing. If it is broadly true that ground-foragers are attracted to the most productive areas within a landscape, yet sensitive to human-related disturbances, it is not surprising that they are declining throughout rural landscapes in Australia (Ford and Howe 1980; Recher and Lim 1990; Robinson 1991; Cale 1993; Paton *et al.* 1994).

As mentioned previously, a portion of the population of both winter and summer visitors was present during the spring survey. During this time, winter visitors were more common at higher altitudes and summer visitors were more common at lower altitudes (Tables 2.4 and 2.6). It is likely that the increased abundance of winter visitors in stringybark woodland at higher altitudes was a response to the flowering of drooping mistletoes (Tables 2.4 and 2.6). Recher (1985) has suggested that in low-fertility habitat where insect pollination is low, nectar may be produced over the winter period to attract honeyeaters from the richer habitats where they breed. If this is so, then winter visitors may be important pollinators of trees, mistletoes and shrubs in stringybark woodland on the Armidale Plateau. To a lesser extent, the increased abundance of summer visitors at lower altitudes may be due to the flowering and fruiting of the box mistletoe. Resident species were also more common at lower altitudes on richer, more productive soils (HMDS only, Tables 2.4 and 2.6).

The abundance of arthropods is related to the nutrient availability and amount of moisture in the soil (Lambert and Turner 1983; Majer *et al.* 1992). Consequently a more diverse assemblage of birds is likely to occur where the soils are relatively moist and nutrient-rich (Recher 1985; Braithwaite *et al.* 1989; Recher and Majer 1994, Stokes in prep.). The natural balance appears to have been upset on the Armidale Plateau, with a large proportion of bird species being more common in the marginal habitat. That is, the degraded stringybark woodland with low vegetation species diversity, that occurs on the drier, poorer soils at higher altitudes. The explanation for this may be historical. The gum-box woodland has been extensively cleared and much of what remains is in small, highly degraded woodland patches. As a result the bird community that was originally associated with this gum-box woodland may no longer be a complete avifauna. As has been mentioned the smallest, highly degraded patches of gum-box woodland (< 2 ha) that are common throughout the Armidale Plateau, are under-represented in the present study. This is why the stringybark woodland patches that were surveyed were generally smaller and more degraded than the gum-box woodland patches surveyed (Table 2.3). The increased abundance of territorial species such as noisy miners and fuscous honeyeaters in the gum-box woodland at lower altitudes (Chapter 3), may also explain why bird species richness was greater in the stringybark woodland. In addition, as has been mentioned, the increased presence of logs in stringybark woodland may result in many bird species finding this woodland type more suitable. The effect of altitude on bird assemblages on the Armidale Plateau is similar to that described by Loyn (1987) for birds in forest fragments in Victoria. In his study, Loyn (1987) found that altitude was positively correlated with forest bird species richness, negatively correlated with farmland species, and negatively associated with noisy miner abundance.

2.6.12 Linear Habitat Features: Water-courses and Strips of Woodland

As well as being more productive, woodland closer to water-courses is likely to provide a relatively moist habitat during the dry winter months. This, and the fact that water-courses seem to act as fly-ways for birds moving through the Armidale Plateau (Chapter 3), may explain why winter visitors were more common on sites close to water-courses. It is also likely that birds were attracted to the relatively heterogeneous and sloping country close to water-courses (Loyn 1985b; Smith 1985). Other studies have demonstrated that land bird species richness is greater where waterways were present within the forest or woodland habitat. Stokes (in prep.) found that red-browed treecreepers were more common along water-courses on the Armidale Plateau. In subtropical rainforest remnants in northeastern New South Wales, Howe *et al.* (1981) found that common species and uncommon species, as well as total bird species richness, increased in remnants with water-courses present. In southeast Queensland, Grover and Slater (1994) found that bird species richness was greater if a water source was present within remnants of *Melaleuca* forest.

Catterall *et al.* (1991) have described how the edges of suburban forest remnants, as well as strips of forest, tend to be dominated by large territorial species, preventing other species from using this habitat. To some extent this appears to be happening on the Armidale Plateau, with the increase in abundance of noisy miners throughout the landscape. A single bird group, the summer visitors, were more common in roadside vegetation. This group of birds includes some hollow-nesters, which were more common in gum-box woodland (although many of the dead trees had been removed from roadsides) and ground-foragers, such as the sacred kingfisher and rufous song-lark, which were associated with tall herbaceous vegetation (Table 2.6 - HMDS). However, as well as using roadside vegetation as habitat, it is also likely that summer visitors use the strips of vegetation along roads as fly-ways (see Chapter 3).

In a landscape where much of the remnant vegetation is reduced to small, isolated, heavily grazed patches, roadside vegetation with its reduced grazing pressure is likely to be an important conservation resource (Bennett 1991; Hibberd and Soutberg 1991; Robinson 1991). This appears to be the case on the Armidale plateau where roadside vegetation often occurs on the flatter, richer soils at lower altitudes, from which most of the original gum-box woodland has been cleared. Furthermore, grazing pressure is slightly reduced and the herb vegetation well developed in the roadside vegetation (Tables 2.3 and 2.4). In the present study, a large proportion (56%) of the bird species that were recorded in woodland patches, also occurred in strips of woodland (n = 26 strip transects). This suggests that strips of woodland were being used by a number of bird species on the Armidale Plateau, either as foraging habitat, or as thoroughfares for movement (see Chapter 3). This situation is similar to many other rural landscapes in Australia, where a large proportion of the bird species known to occur in patches of remnant habitat also occur in strips of roadside vegetation (Arnold and Weeldenburg 1990; Cale 1990; Lynch and Saunders 1991; Leach and Hines 1993).

2.6.13 The Armidale Plateau: Management

It is now generally accepted that a stable equilibrium state is not a fundamental property of ecological systems (DeAngelis and Waterhouse 1987). As has been argued by Hobbs and Huenneke (1992), natural disturbance regimes may be desirable, but are often impracticable in the altered settings of contemporary reserves. They further state that there is a need for managers to acknowledge the actual disturbance regime operating currently in remnants, and to take an active role in designing the disturbance regime, tailoring it to the landscape, the biotic community, and to specific conservation goals.

The response of bird groups to moderate levels of disturbance suggests that a similar strategy may be appropriate for conserving bird species on the Armidale Plateau.

It appears that a large portion of the bird species are utilising the moderately disturbed woodland which occurs between the existing conservation reserves, state forests and between the few large patches of undisturbed woodland on private land. The careful management of these 'off-reserve' areas by landholders may be essential to halt the decline of bird species on the Armidale Plateau. If further clearing of woodland is discouraged and existing large areas of woodland protected, it is likely that a large proportion of the bird species could be conserved throughout the Armidale Plateau within a network of inter-connected remnants each of which is at least 20 ha in area. Gum-box woodland is presently under-represented, so regeneration of existing gum-box woodland, and the creation of new woodlots on the richer country at lower altitude, should be given priority. Having established this heterogeneous remnant network, bird species richness can be maximised by maintaining habitat heterogeneity within each woodland patch. As a general guide, bird species richness will be greater in a patch of woodland on the Armidale Plateau if: 1) it is at least 20 ha in area, 2) it is does not become increasingly isolated from other woodlots, 3) there is a range of tree and shrub species present, 5) grazing is moderate so that tall herbaceous vegetation, as well as patches of dense understorey vegetation are present, 6) mistletoes are present, 7) large trees with nesting-hollows are present, 8) fallen timber is left to break down naturally, 9) there is a water-course within or near the woodlot.

This management approach takes an intermediate pathway between conserving species richness and conserving individual species. The guidelines presented here for 'off-reserve' areas apply to the bird community as a whole and to groups of species, rather than to all individual species. There are some bird species, particularly those that depend on extensively wooded areas, that may disappear from the woodland areas between large reserves. However, there is a strong tendency for such species to be marginal to the New England Tablelands (Chapter 4). There is good reason to expect that the majority of bird species that occur on the Armidale Plateau are threatened with local if not widespread extinction if the current level and intensity of land-use continues. Under such circumstances few would disagree that it is valid to search for general patterns in the bird assemblages, with a view to developing management guidelines for maintaining local species richness. This is what has been proposed in the present study for birds on the Armidale Plateau With proper support by government and industry, these conservation goals should be achievable by the majority of landholders. It is our good fortune that Australian ecosystems are often forgiving, despite their fragility. The increasing awareness and willingness of landholders and agricultural advisers to adopt landcare measures is also reason for hope.

2.6.14 In Summary:-

1. Birds were surveyed for 20 minutes, and 35 environmental features were described in 294 woodland sites. In a subset of 63 sites, birds were surveyed each season for two years.

2. Data were analysed using three kinds of multivariate analysis - a) Stepwise regression using principal components, b) Standard multiple regression using actual environmental variables and c) Hybrid multidimensional scaling.

3. The most frequently selected predictors of bird abundance were the percent dieback, number of mistletoes, altitude, presence of large trees, dominant tree species, area of the woodland patch, and the number of logs.

4. The cumulative number of bird species from the eight seasonal surveys over two years indicate that bird species richness per 1.2 ha site was greatest in the largest woodland patches

(> 400 ha). However, the single survey in 294 sites indicates that bird species richness was negatively correlated with area, and that most bird groups were less common in larger woodland patches. This single survey also indicates that bird species richness was greatest in woodland patches that were greater than six hectares yet smaller than 400 ha.

5. Area and isolation effects were stronger in woodland sites where noisy miners were present.

6. Smaller woodland patches were often degraded by eucalypt dieback, timber removal and heavy grazing pressure, resulting in a reduction of the tree canopy and understorey vegetation. Nest predators and species which have a higher population density, such as the noisy miner, were more common in these smaller woodland sites. In these small woodland patches the herb vegetation was well developed, which favoured ground-foraging and open-country species.

7. A number of bird groups were more common where there was reduced grazing pressure, greater vegetation biomass, greater vegetation species richness, larger trees and fallen timber. Such groups included:- honeyeaters, fruit-eaters, understorey-

dependent species, winter visitors, bark-foragers, and hollow-nesters. These environmental features were generally associated with larger patches of woodland that were further from human settlement.

8. The gum-box woodland at lower altitude has been most extensively cleared for grazing, because it occurs on the richer soils. Uncommon bird species, species with a low population density, hollow-nesters and resident species were all more common in larger patches of gum-box woodland at lower altitudes. These groups of species are most vulnerable to current land management on the New England Tablelands. Ground-foragers also appear to be vulnerable, because they are sensitive to habitat disturbance and associated with small patches of gum-box woodland at lower altitude.

9. Bird species richness was greatest in patches of stringybark woodland, on the poorer soils at higher altitudes. This may be partly historical - the extensive clearing and degradation of gum-box woodland is likely to have reduced the bird community associated with this woodland type to such a state, that it is no longer a complete avifauna. It is also likely that the increased abundance of territorial bird species in the gum-box woodland at lower altitudes results in the other bird species being forced into the marginal stringybark habitat. The removal of timber from the gum-box woodland sites is also likely to have had an impact on bird species richness.

10. Over half of the bird species that were recorded in patches of woodland (56%), also occurred in strips of woodland (most of which were along roadsides).

Figure 2.1: Distribution of 294 census sites in woodland patches within a 60 km of Armidale, on the New England Tablelands, Northeast New South Wales (Data from NSW NPWS).

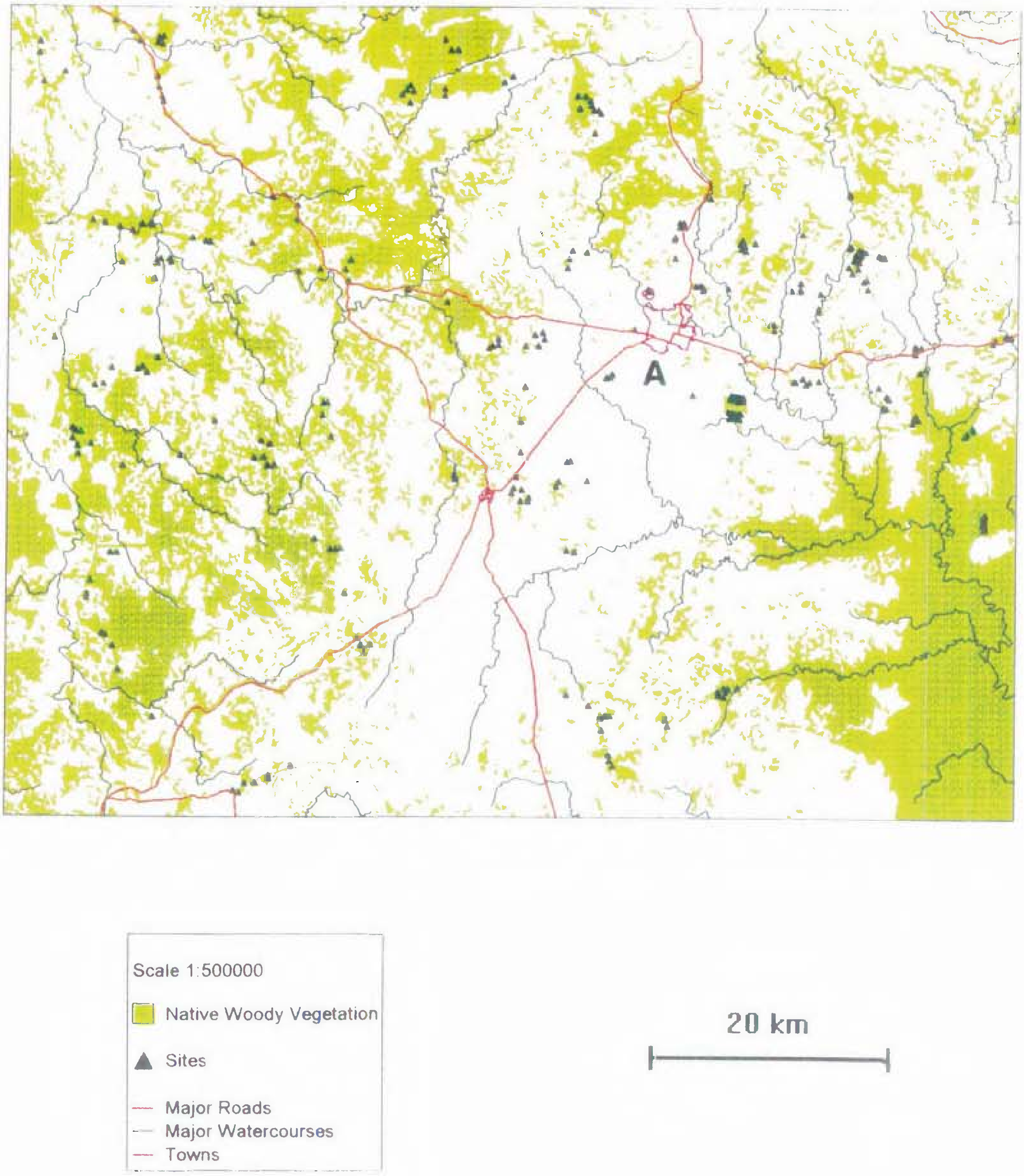
A
Scale 1:500000
🟩 Native Woody Vegetation
▲ Sites
Major Roads
Major Watercourses
Towns
20 km

Figure 2.2: The effect of area of woodland patch and presence of noisy miners on the mean number of bird species recorded during a single 20 minute census (±SE). The number of replicate transects in each size category are as follows: > 400 ha (all sites = 77; miner site; n = 5, non-miner site n = 72), 101-400 ha (all sites = 79; miner site; n = 5, non-miner site n = 74), 21-100 ha (all sites = 82; miner site; n = 17, non-miner site n = 65), 6-20 ha (all sites = 35; miner site; n = 15, non-miner site n = 20), < 6 ha (all sites = 21; miner site; n = 7, non-miner site n = 14) and strips of woodland (all sites = 26; miner site; n = 5, non-miner site n = 21). Mean number of species differed significantly in all sites (H = 10.81, d.f. = 4, $p < 0.05$), on sites with miners (H = 12.1, d.f. = 4, $p < 0.05$) and on sites without miners (H = 21.96, d.f. = 4, $p < 0.01$). Figure modified from Ford *et al.* in press.

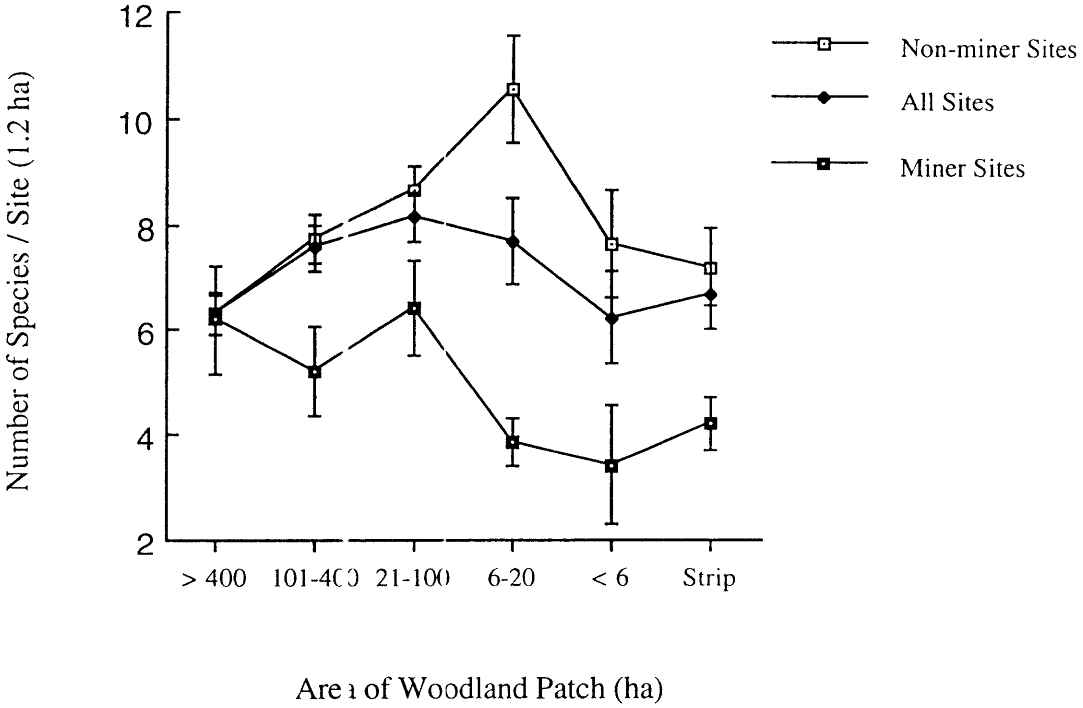
Number of Species / Site (1.2 ha)
12
10
8
6
4
2
> 400
101-400
21-100
6-20
< 6
Strip
Area of Woodland Patch (ha)
Non-miner Sites
All Sites
Miner Sites

Figure 2.3: The effect of area of woodland patch on the mean number of bird species recorded during a single 20 minute census in stringybark woodland and gum-box woodland (±SE). The number of replicate transects in each size category are as follows: > 400 ha (stringybark, n = 16; gum-box, n = 28), 101-400 ha (stringybark, n = 35; gum-box, n = 19), 21-100 ha (stringybark, n = 37; gum-box, n = 27), 6-20 ha (stringybark, n = 15; gum-box, n = 15), < 6 ha (stringybark, n = 11; gum-box, n = 8). There were no significant area effects (stringybark, F = 2.1, d.f. = 4, 109, p = 0.08; gum-box, F = 1.2, d.f. = 4, 92, p = 0.33). Noisy miner sites were included in the analysis.

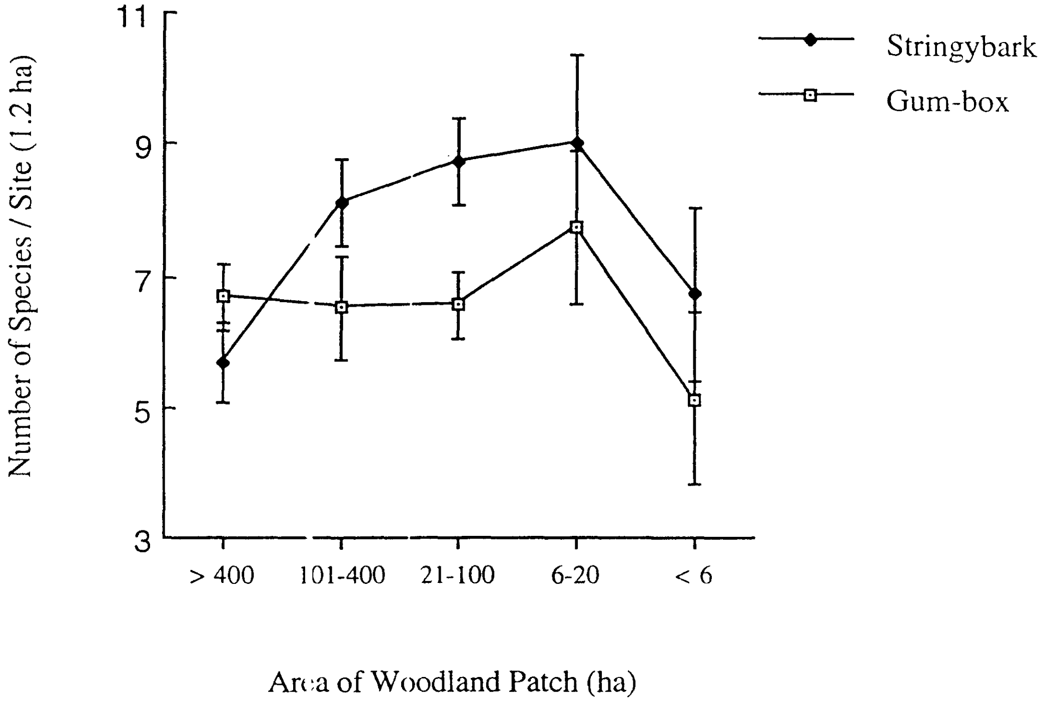
Number of Species / Site (1.2 ha)
11
9
7
5
3
> 400
101-400
21-100
6-20
< 6
Area of Woodland Patch (ha)
Stringybark
Gum-box

Figure 2.4: Cumulative number of bird species at sites in woodland patches that were > 400 ha (n = 12, includes no miner sites), 101 - 400 ha (n = 13, includes 1 miner site), 21 - 100 ha (n = 24, includes 7 miner sites); 6 - 20 ha (n = 6, includes 3 miner sites) and < 6 ha (n = 8, includes 2 miner sites). Sites with noisy miners were included in the analysis. These were sites in which noisy miners were present in at least five of the eight surveys. There were no significant area effects on the cumulative number of bird species (Kruskal Wallis: $H = 5.5$, d.f. = 4, $p = 0.2$).

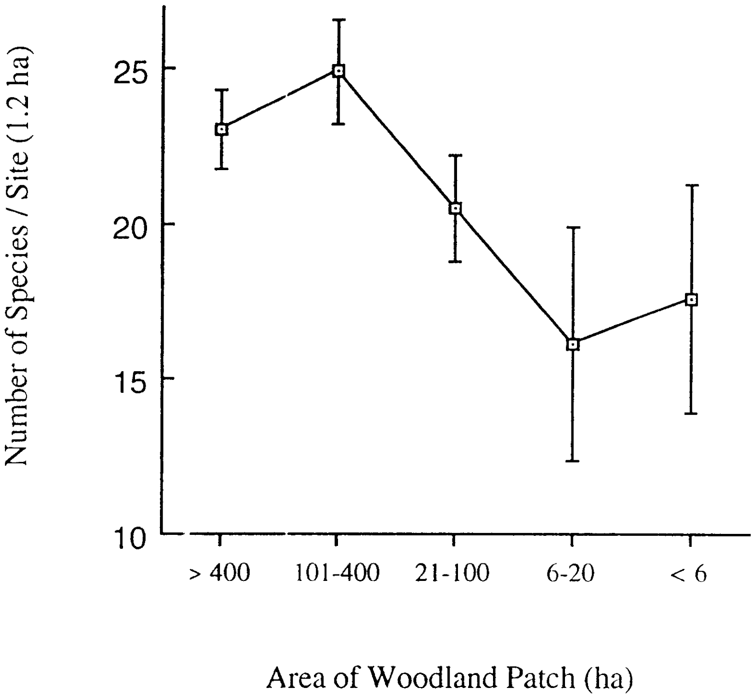
Number of Species / Site (1.2 ha)
25
20
15
10
> 400
101-400
21-100
6-20
< 6
Area of Woodland Patch (ha)

Table 2.1: Bird species groupings: 1) # sites recorded out of 294 sites. Common species are defined as occurring in at least 65 of the 294 sites (10 spp.) and uncommon species defined as occurring in fewer than 10 sites (39 spp.). 2) species with low population density (< 1.3 indiv./ha, 11 spp.), 3) species with high population density (> 2.5 indiv./ha, 11 spp.), 4) woodland species (remnant dependent, 74 spp.), 5) Open-country species (14 spp.), 6) Ground-foragers (excl. open-country spp., 16 spp.), 7) understorey species (7 spp.), 8) bark-foragers (7 spp.), 9) foliage-gleaners and snatchers (excl. honeyeaters, 13 spp.), 10) fruit-eaters (excl. open-country spp., 6 spp.), 11) honeyeaters (excl. noisy miners, 13 species), 12) nest predators (7 spp.), 13) hollow-nesters (excl. open-country spp., 13 spp.). 14) summer visitors (excl. open-country spp., 22 spp.), 15) winter visitors (excl. open-country spp., 6 spp.), 16) residents (excl. open-country spp., 36 spp.). * species not included in analysis of repeated measures in 63 sites.

SPECIES	CODE:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Brown Goshawk	*Accipiter fasciatus*	5	•		•												•
Little Eagle	*Hieraaetus morphnoides*	2			•												•
Peaceful Dove *	*Geopelia striata*	2			•		•										•
Common Bronzewing *	*Phaps chalcoptera*	1			•		•										•
Crested Pigeon	*Ocyphaps lophotes*	1				•											
Galah	*Cacatua roseicapilla*	7				•											
Sulphur-crested Cockatoo *	*Cacatua galerita*	2				•								•			
Musk Lorikeet	*Glossopsitta concinna*	9		•	•									•			
Little Lorikeet *	*Glossopsitta pusilla*	6			•									•			
King Parrot *	*Alisterus scapularis*	1			•						•			•			•
Crimson Rosella	*Platycercus elegans*	76			•						•			•			•
Eastern Rosella	*Platycercus eximius*	84		•		•											
Red-rumped Parrot	*Psephotus haematonotus*	1				•											
Pallid Cuckoo	*Cuculus pallidus*	5			•										•		
Fan-tailed Cuckoo	*Cuculus pyrrhophanus*	2			•					•					•		
Horsfield's Bronze Cuckoo *	*Chrysococcyx basalis*	3			•										•		
Golden Bronze Cuckoo	*Chrysococcyx lucidus plagosus*	11			•					•					•		
Channel-billed Cuckoo *	*Scythrops novaehollandiae*	1			•										•		
Owlet Nightjar *	*Aegotheles cristatus*	1			•		•							•			•
Laughing Kookaburra	*Dacelo gigas*	12			•		•						•	•			•
Sacred Kingfisher	*Halcyon sancta*	23			•		•							•	•		
Rainbow Bee-eater	*Merops ornatus*	2													•		
Dollarbird	*Eurystomus orientalis*	4	•		•									•	•		
Welcome Swallow	*Hirundo neoxena*	4				•											
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	50			•					•					•		
Little Cuckoo-shrike	*Coracina papuensis*	2			•					•							
White-winged Triller	*Lalage sueurii*	3			•					•					•		
Scarlet Robin	*Petroica multicolor*	22			•		•										•
Hooded Robin	*Melanodryas cucullata*	3	•		•		•										•

Table 2.1 continued

SPECIES	CODE:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Eastern Yellow Robin	*Eopsaltria australis*	24			•		•										•
Jacky Winter	*Microeca leucophaea*	14			•												
Crested Shrike-tit	*Falcunculus frontatus*	16	•		•				•								•
Rufous Whistler	*Pachycephala rufiventris*	102			•					•					•		
Golden Whistler	*Pachycephala pectoralis*	14	•		•					•						•	
Grey Shrike-thrush	*Colluricincla harmonica*	46	•		•		•		•				•				•
Restless Flycatcher	*Myiagra inquieta*	10	•		•												•
Leaden Flycatcher	*Myiagra rubecula*	38			•										•		
Satin Flycatcher	*Myiagra cyanoleuca*	1			•										•		
Grey Fantail	*Rhipidura fuliginosa*	76			•			•							•		
Willie Wagtail	*Rhipidura leucophrys*	46				•											
Rufous Song-lark	*Cinclorhamphus mathewsi*	12				•									•		
Superb Fairy Wren	*Malurus cyaneus*	30		•	•		•	•									•
White-browed Scrub Wren	*Sericornis frontalis*	12			•												•
Speckled Warbler	*Sericornis sagittatus*	13	•		•		•										•
Broad-tailed Thornbill *	*Acanthiza apicalis*	1			•					•							
Brown Thornbill	*Acanthiza pusilla*	26			•			•									•
Buff-rumped Thornbill	*Acanthiza reguloides*	75			•												•
Striated Thornbill	*Acanthiza lineata*	115		•	•					•							•
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	34				•											
White-throated Warbler	*Gerygone olivacea*	56	•		•					•					•		
Weebill	*Smicrornis brevirostris*	9			•					•							
Varied Sittella	*Daphoenositta chrysoptera*	48			•				•								•
White-throated Treecreeper	*Climacteris leucophaea*	77			•				•					•			•
Red-browed Treecreeper	*Climacteris erythrops*	6			•				•					•			•
Brown Treecreeper	*Climacteris picumnus*	27			•		•		•					•			•
Red Wattlebird	*Anthochaera carunculata*	50			•							•					
Noisy Friarbird	*Philemon corniculatus*	81			•							•			•		
Noisy Miner	*Manorina melanocephala*	62		•	•												•
Yellow-faced Hon.	*Lichenostomus chrysops*	55			•			•				•				•	

Table 2.1 continued

SPECIES	CODE:	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Yellow-tufted Hon. *	*Lichenostomus melanops*	1			●							●					
White-eared Hon.	*Lichenostomus leucotis*	21			●			●	●			●					●
Fuscous Hon.	*Lichenostomus fuscus*	83		●	●							●					●
White-plumed Hon. *	*Lichenostomus pencillatus*	10		●	●							●					●
Black-chinned Hon. *	*Melithreptus gularis*	2			●							●					
Brown-headed Hon.	*Melithreptus brevirostris*	17			●							●					●
White-naped Hon.	*Melithreptus lunatus*	71			●							●				●	
Tawny-crowned Hon.	*Phylidonyris melanops*	1			●							●					
Eastern Spinebill	*Acanthorhynchus tenuirostris*	22			●							●				●	
Mistletoebird	*Dicaeum hirundinaceum*	38			●						●				●		
Spotted Pardalote	*Pardalotus punctatus*	29	●		●					●						●	
Striated Pardalote	*Pardalotus striatus*	68			●					●				●			●
Silvereye	*Zosterops lateralis*	4			●						●	●				●	
European Goldfinch *	*Carduelis carduelis*	1				●											
Red-browed Firetail	*Emblema temporalis*	5			●		●	●									●
Diamond Firetail	*Emblema guttata*	4			●		●										●
Double-barred Finch	*Poephila bichenovii*	3			●		●	●									●
Common Starling	*Sturnus vulgaris*	21		●		●											
Olive-backed Oriole	*Oriolus sagittatus*	10	●		●						●				●		
Satin Bowerbird *	*Ptilonorhynchus violaceus*	2			●						●						●
White-winged Chough	*Corcorax melanorhamphus*	30		●	●		●										●
Australian Magpie Lark	*Grallina cyanoleuca*	9				●											
White-br. Woodswallow *	*Artamus superciliosus*	6		●	●										●		
Dusky Woodswallow	*Artamus cyanopterus*	33			●										●		
Grey Butcherbird	*Cracticus torquatus*	37			●								●				●
Pied Butcherbird *	*Cracticus nigrogularis*	1			●								●				●
Australian Magpie	*Gymnorhina tibicen*	54				●							●				
Pied Currawong	*Strepera graculina*	35			●								●		●		
Australian Raven *	*Corvus coronoides*	5				●							●				

Table 2.2: The mean values of the 35 environmental variables (n = 294 sites). The transformations given are those used to improve normality prior to standardization for the regression analyses.

Variable	Mean (± SD) for 294 sites	Transformation
% Canopy Cover (>8m)	18.12 ± 7.6	arcsine
% Shrub Cover (<2m)	7.93 ± 9.5	log
% Sub-can. Cover	7.38 ± 5.1	sqrt
Vegetation Biomass	33.43 ± 14.2	sqrt
% Herb Cover	54.38 ± 24.2	arcsine
Herb Height (cm)	11.79 10.9	log
Canopy Height (m)	17.49 ± 3.2	arcsine
Tree Diameter (cm)	30.11 ± 7.3	none
Largest Tree (dbh x ht)	1301.2 ± 454.5	sqrt
# Canopy Spp.	3.08 ± 1.1	log
# Sub-canopy Spp.	1.87 ± 0.8	log
# Shrub Spp.	4.12 ± 2.4	sqrt
% Acacia Shrub	7.55 ± 11.9	sqrt
Total # Veg. Spp.	9.07 ± 3.3	none
Propn Gum Trees	0.23 ± 0.3	sqrt
Propn Box Trees	0.25 ± 0.3	sqrt
Propn Gum-box	0.48 ± 0.4	log
Propn Stringybark	0.52 ± 0.4	log
Tree Density (# per 1.2 ha)	41.44 ± 25.6	sqrt
# Drooping Mistletoe	2.54 ± 4.5	sqrt
# Box Mistletoe	2.92 ± 5.0	sqrt
Total # Mistletoes	5.70 ± 6.8	sqrt
# Logs	4.61 ± 2.7	sqrt
# Cut Stumps	4.53 ± 3.4	arcsine
% Dieback	31.88 ± 18.6	sqrt
# Dead Trees	4.37 ± 4.9	log
Slope	1.31 ± 0.7	none
Altitude (m)	972.47 ± 134.6	none
Area of Patch (ha)	213.89 ± 199.6	log
Isolation	0.55 ± 0.1	sqrt
Dist. Buildings (km)	1.50 ± 1.3	log
Dist. Perm. Water (km)	0.55 ± 0.4	sqrt
Dist Water-course (km)	0.26 ± 0.3	sqrt
Dist Road (km)	0.78 ± 0.8	sqrt
Dist. Edge (km)	0.22 ± 0.2	sqrt

Table 2.3: Spearman rank correlation matrix for selected environmental variables using estimation sample (n = 147, *p < 0.05, **p < 0.01, ***p < 0.001).

Variable	1	2	3	4	5	6	7	8	9	10	11	12	13	14
1. % Canopy Cover	1.0													
2. % Shrub Cover	0.07	1.0												
3. Veg. Biomass	0.64***	0.67***	1.0											
4. % Herb Cover	–0.19*	–0.35***	–0.35***	1.0										
5. Herb Height	–0.17*	–0.07	–0.14	0.39***	1.0									
6. Canopy Height	0.41***	0.04	0.22**	0.05	–0.04	1.0								
7. Tree Diameter	–0.04	–0.10	–0.09	0.20*	0.00	0.31***	1.0							
8. Largest Tree	0.17*	–0.06	0.08	0.10	–0.06	0.47***	0.39***	1.0						
9. % Acacia Shrubs	0.27***	0.24**	0.21**	–0.15	–0.04	0.35***	0.03	0.09	1.0					
10. # Shrub Spp.	0.14	0.60***	0.33***	–0.40***	–0.02	0.20*	–0.08	0.03	0.54***	1.0				
11. Total # Veg. Spp.	0.19*	0.54***	0.39***	–0.34***	0.02	0.22**	–0.07	0.05	0.57***	0.90***	1.0			
12. Propn Gum/Box	–0.12	0.04	–0.07	0.19*	0.32***	–0.02	–0.12	0.03	–0.01	0.15	0.2*	1.0		
13. Propn Stringybark	0.12	–0.04	0.07	–0.19*	–0.32***	0.02	0.12	–0.03	0.01	–0.15	–0.2*	–1.0***	1.0	
14. Tree Density	0.56***	–0.02	0.27***	–0.34***	–0.24**	0.09	–0.12	0.00	0.16*	0.19*	0.19*	–0.34***	0.34***	1.0
15. Total # Mistletoes	–0.20*	0.06	–0.11	–0.08	–0.11	0.11	–0.03	0.13	0.12	0.18*	0.22**	0.02	–0.02	–0.13
16. # Logs	0.27***	–0.11	0.13	–0.11	–0.28***	0.13	0.10	0.12	0.19*	0.07	0.06	–0.33***	0.33***	0.39***
17. # Cut Stumps	0.09	–0.10	0.02	0.02	–0.04	0.20*	0.20*	0.12	0.12	–0.10	–0.09	–0.35***	0.35***	0.09
18. % Dieback	–0.20*	–0.23**	–0.24**	0.17*	0.21**	–0.09	0.27***	–0.05	–0.08	–0.18*	–0.15	–0.19*	0.19*	0.06
19. # Dead Trees	–0.02	–0.06	–0.05	0.01	0.00	–0.21**	0.01	–0.18*	–0.01	0.02	0.05	–0.01	0.01	0.28***
20. Slope	–0.01	0.06	0.11	–0.17*	0.01	–0.13	–0.06	–0.26**	0.09	0.01	0.09	–0.23**	0.23**	0.04
21. Altitude	0.06	–0.11	–0.01	0.20*	–0.06	0.17*	0.36***	0.19*	0.06	–0.13	–0.19*	–0.26**	0.26**	0.12
22. Area Patch	0.34***	0.13	0.24**	–0.26**	–0.09	0.13	–0.07	–0.07	0.28***	0.44***	0.52***	0.07	–0.07	0.36***
23. Isolation	–0.23**	–0.21**	–0.19*	0.08	0.00	–0.01	0.10	–0.09	–0.17*	–0.33***	–0.28***	–0.09	0.09	–0.17*
24. Dist. Buildings	0.16*	0.01	0.06	–0.10	0.00	–0.04	–0.16*	–0.16*	–0.01	0.20*	0.16*	–0.03	0.03	0.26**
25. Dist. Perm. Water	0.20*	0.24**	0.26**	–0.16*	0.01	0.10	–0.03	0.09	0.00	0.33***	0.28***	0.12	–0.12	0.18*
26. Dist. Water-course	–0.11	0.02	–0.10	–0.06	–0.06	0.06	0.13	0.13	0.03	0.04	–0.03	0.00	0.00	–0.14
27. Dist. Road	0.08	–0.01	0.07	–0.24**	–0.37***	–0.07	–0.03	–0.14	–0.14	–0.05	–0.04	–0.35***	0.35***	0.32***
28. Dist. Edge	0.22**	0.05	0.16*	–0.08	–0.13	0.16*	0.09	0.01	0.09	0.25**	0.27***	–0.11	0.11	0.25**

Table 2.3 continued

Variable	15	16	17	18	19	20	21	22	23	24	25	26	27	28
15. Total # Mistletoes	1.0													
16. # Logs	–0.14	1.0												
17. # Cut Stumps	0.22**	0.17*	1.0											
18. % Dieback	–0.10	–0.10	0,12	1.0										
19. # Dead Trees	–0.24**	0.14	–0.17*	0.24**	1.0									
20. Slope	0.08	0.09	0.07	0.05	0.00	1.0								
21. Altitude	–0.06	0.27***	0.15	0.13	0.11	0.03	1.0							
22. Area Patch	–0.06	0.28***	–0.17*	–0.11	0.17*	0.08	–0.16*	1.0						
23. Isolation	0.17*	–0.21**	0.14	0.23**	–0.12	0.21**	–0.04	–0.26**	1.0					
24. Dist. Buildings	-0.27***	0.25**	-0.17*	-0.05	0.19*	–0.10	–0.20*	0.41***	–0.30***	1.0				
25. Dist. Perm. Water	–0.22**	0.11	–0.23**	–0.11	0.13	–0.11	–0.07	0.47***	–0.27***	0.41***	1.0			
26. Dist. Water-course	0.07	–0.10	0.20*	–0.05	–0.09	–0.18*	0.04	–0.08	0.05	–0.13	0.03	1.0		
27. Dist. Road	–0.09	0.19*	–0.15	0.13	0.17*	0.22**	0.04	0.26**	0.17*	0.23**	0.09	–0.21**	1.0	
28. Dist. Edge	–0.16*	0.23**	–0.15	–0.04	0.11	0.09	–0.02	0.61***	–0.13	0.41***	0.39***	–0.04	0.31***	1.0

Table 2.4: Summary of site characteristics and associated bird groups. Groupings are based on Table 2.3 and Appendix 1.

Environmental Features		Bird Groups
Sites in Large Patches	Higher Altitude	Large Patch / Higher Altitude
• Generally at Lower Altitude • Away from Edge of Patch • Away from Farm Buildings • Away from Permanent Water (farm dams) • Away from Roads (less accessible) • Fewer Cut Stumps • More Dead Trees (not removed) • More Logs (not removed for fire-wood) • Greater Tree Density • Increased Vegetation Biomass (understorey and canopy) • Greater Vegetation Species Richness • Increased Percentage *Acacia* spp. in the Shrub Vegetation indicating lower Grazing Pressure • Dieback Not Severe	• Mostly Stringybark Woodland • Rocky Nutrient-poor Soils • Sloping Country • Greater Tree Density • Larger Average Tree Diameter • Taller Trees • Greater Canopy Cover • Reduced Herbaceous Vegetation • Reduced Vegetation Species Richness	• Woodland-dependent Spp. • Understorey-dependent Spp. • Fruit-eaters • Honeyeaters • Winter Visitors
	Lower Altitude	Large Patch / Lower Altitude
	• Mostly Gum-Box woodland • Nutrient-rich Soils • On Flatter Country • Smaller Average Tree Diameter • Reduced Tree Density • Reduced Canopy Cover • Greater Tree and Shrub Species Richness • Greater Vegetation Biomass (Mostly Understorey) • More Extensive Herbaceous Vegetation	• Uncommon Spp. • Low Population Density Spp. • Bark-foragers (also in stringybark woodland at higher altitude) • Hollow-nesters • Resident Spp.
Sites in Small Patches	Higher Altitude	Small Patch / Higher Altitude
• Generally at Higher Altitude • Isolated from Other Woodland • Closer to Farm Buildings • Closer to Permanent Water (farm dams) • Closer to Roads • Closer to the Edge of the Patch • Dieback More Extensive • More Cut Stumps • Reduced Vegetation Biomass • Herbaceous Vegetation More Extensive	• Stringybark Woodland • Large, Tall Trees • Drooping Mistletoe Present • Sloping Country, Nutrient-poor Rocky Soils • Reduced Herbaceous Vegetation • Logs Present	• # Bird Species • Simpson's Diversity Index • Common Spp. • Foliage-gleaners • Nest Predators
	Lower Altitude	Small Patch / Lower Altitude
	• Gum-Box Woodland • Many Mistletoes (mostly box mistletoe) • More Likely to be Isolated • Smaller Average Tree Diameter • Close to Roads • Logs Removed • Extensive, Tall Herbaceous Vegetation • Flatter Country	• # Birds • Open-country Spp. • High Population Density Spp. • Summer Visitors (also visit stringybark) • Ground-foragers

Table 2.5: Principal components analysis of the 35 environmental variables. Only coefficients greater than 0.5 ($P < 0.001$, $n = 294$) are included.

Variable * untransformed	PC1	PC2	PC3*	PC4	PC5*	PC6	PC7	PC8	PC9	PC10*	PC11
% Canopy Cover (>8m)						0.772					
% Shrub Cover (<2m)					0.638						
% Sub-can. Cover					0.836						
Veg Biomass					0.856						
% Herb. Cover										0.510	
Dom. Herb. Ht (cm)										0.801	
Dom. Canopy Ht (m)				0.793							
Tree Diameter (cm) *				0.760							
Largest Tree (dbh x ht)				0.741							
# Sub-canopy Spp.	0.630										
# Shrub Spp.	0.785										
% Acacia Shrubs	0.809										
Total # Veg. Spp. *	0.824										
Propn Gum Trees		-0.900									
Propn Gum-box		-0.825									
Propn Stringybarks		0.796									
Tree Density (# per 1.2 ha)						0.785					
# Drooping Mistletoe			0.740								
# Box Mistletoe			0.591								
Total # Mistletoes			0.924								
% Dieback								0.783			
Slope *											0.644
Altitude (m) *									0.759		
Area of Patch (ha)							0.517				
Dist. Buildings (km)							0.657				
Dist. Perm. Water (km)							0.762				
Dist Water-course (km)											-0.677
Dist. Edge (km)							0.727				
% Variance	9.08	9.01	6.02	6.78	6.47	6.78	8.25	4.15	4.29	4.21	4.49
Cumulative Variance	9.08	18.09	24.11	30.89	37.36	44.14	52.39	56.54	60.83	65.04	69.53

Table 2.6: Environmental variables that were significant predictors using; stepwise multiple regression to select principal components (PCA), standard multiple regression from a single spring survey (SMR - Spring 1990) and standard multiple regression from repeated surveys (SMR - seasonal) The 'most likely predictors' selected using hybrid multidimensional scaling (HMDS) have no measure of significance. See Appendices 2 to 5 for the variable coefficients, model constants and R^2 (ADJ) values for specific models. Where no variables were selected this is indicated (None). Where the analysis was inappropriate or the model rejected this is also indicated (-).

Bird group	PCA	SMR - Spring 1990	SMR - Seasonal	HMDS
# Species (+ve)	PC3 (Mistletoes) PC2 (Stringybark) PC4 (Large Trees)	# Logs % Herb Cover	None	Altitude Propn Stringybark Veg. Biomass # Logs
# Species (-ve)	PC8 (Dieback) PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings, Area)	Dist. Perm. Water Dieback	Dieback Slope	Propn Gum-Box Area
Simpson's I (+ve)	PC3 (Mistletoes) PC2 (Stringybark) PC6 (Tree Density, % Canopy Cover)	# Logs	-	-
Simpson's I (-ve)	PC8 (Dieback) PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings Area)	Dist. Water-course Dieback Dist. Perm. Water	-	-
# Birds (+ve)	PC3 (Mistletoes) PC4 (Large Trees) PC10 (% Herb Cover, Herb Height)	# Mistletoes % Herb Cover Cut Stumps	-	-
# Birds (-ve)	PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings Area) PC8 (Dieback) PC1 (# Veg. Species, % Acacia)	Dist. Water-course Slope # Veg. Species Dieback	-	-
Uncommon Spp. (+ve)	None	Propn Box	% Acacia	-
Uncommon Spp. (-ve)	PC8 (Dieback) PC9 (Altitude)	Altitude	None	-
Common Spp. (+ve)	PC4 (Large Trees) PC2 (Stringybark) PC6 (Tree Density, % Canopy Cover) PC3 (Mistletoe) PC9 (Altitude)	# Cut Stumps # Logs	None	Altitude Propn Stringybark # Logs
Common Spp. (-ve)	PC8 (Dieback) PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings. Area)	Dist. water-course Dieback	Dieback	Propn Gum-Box Propn Gum Dieback
Low Popn Den. (+ve)	-	# Logs % Herb Cover	None	Propn Stringybark # Logs # Cut Stumps
Low Popn Den. (-ve)	-	Altitude Dist. Building	Dieback	Propn Gum-Box Slope Dieback
High Popn Den. (+ve)	PC10 (% Herb Cover, Herb Height)	% Herb Cover # Dead Trees	Herb height Dist. Building	Propn Gum-Box Propn Gum
High Popn Den. (-ve)	PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings. Area) PC1 (# Veg. Species, % Acacia)	Altitude Dist. Edge	Area	Altitude Propn Stringybark # Logs Dist Edge

Table 2.6 continued

Bird group	PCA	SMR - Spring 1990	SMR - Seasonal	HMDS
Woodland-dependent (dp) Spp. (+ve)	PC3 (Mistletoes)	% Acacia	None	Altitude Propn Stringybark # Logs Veg. Biomass Dist. Edge
Woodland-dp Spp. (-ve)	PC8 (Dieback)	Altitude Isolation	Dieback	Propn Gum-Box
Open-country Spp. (+ve)	PC8 (Dieback) PC10 (% Herb Cover, Herb Height)	% Herb Cover	None	Altitude # Cut Stumps
Open-country Spp. (-ve)	PC1 (# Veg. Species, % Acacia) PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings, Area) PC2 (Stringybark) PC6 (Tree Density, % Canopy Cover)	Dist. Edge	Area Slope	Area # Shrub Species Dist. Edge # Veg. Species
Ground-foragers (+ve)	-	-	None	Tree Density Herb Height Box Mistletoe
Ground-foragers (-ve)	-	-	Dieback Propn Gum	Altitude % Shrub Cover Dieback
Understorey Spp. (+ve)	PC5 (% Shrub Cover, % Sub-can. Cover, Veg. Biomass) PC9 (Altitude) PC3 (Mistletoes) PC2 (Stringybark) PC1 (# Veg. Species, % Acacia)	# Droop. Mistletoe % Shrub Cover	# Sub-can. Spp.	Dist. Buildings # Shrub Species Altitude Dist. Perm. Water Area Propn Stringybark
Understorey Spp. (-ve)	PC8 (Dieback)	None	Dieback Propn Box	None
Bark Foragers (+ve)	PC4 (Large Trees) PC1 (# Veg. Species, % Acacia)	# Logs Canopy Height	% Acacia	# Logs Tree Density # Mistletoes Altitude
Bark-foragers (-ve)	None	None	Dieback	Propn Stringybark Propn Gum-Box
Foliage-gleaners (+ve)	PC2 (Stringybark) PC3 (Mistletoes)	# Mistletoes Veg. Biomass # Logs	None	# Logs Propn Stringybark Tree Density
Foliage-gleaners (-ve)	PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings, Area)	# Veg. Species	Dieback	Altitude Propn Gum-Box Propn Gum
Fruit-eaters (+ve)	PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings, Area) PC4 (Large Trees) PC6 (Tree Density, % Canopy Cover)	Dist. Edge Veg. Biomass # Logs	# Mistletoes	-
Fruit-eaters (-ve)	PC8 (Dieback)	None	Dieback	-

Table 2.6 continued

Bird group	PCA	SMR - Spring 1990	SMR - Seasonal	HMDS
Honeyeaters (+ve)	PC1 (# Veg. Species, % Acacia) PC4 (Large Trees) PC5 (% Shrub Cover, % Sub-can. Cover, Veg. Biomass) PC8 (Dieback) PC6 (Tree Density, % Canopy Cover) PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings, Area) PC3 (Mistletoes) PC9 (Altitude)	# Veg. Species # Dead Trees Veg. Biomass Canopy Height # Droop. Mistletoe Propn Gum-Box	# Mistletoes	Altitude Propn Stringybark # Droop. Mistletoe
Honeyeaters (-ve)	PC2 (Stringybark) PC11 (Dist. Water-course, Flat Country)	None	Dieback	Propn Gum-Box Propn Gum # Box Mistletoe
Nest Predators (+ve)	None	-	None	Altitude
Nest Predators (-ve)	PC7 (Dist. Perm. Water, Dist. Edge, Dist. Buildings, Area)	-	Area	# Shrub Species # Veg. Species Area Dist. Buildings
Hollow-nesters (+ve)	PC4 (Large Trees) PC10 (% Herb Cover, Herb Height) PC1 (# Veg. Species, % Acacia) PC6 (Tree Density, % Canopy Cover)	% Acacia # Logs Dist. Building	Propn Gum-Box % Can. Cover	-
Hollow-nesters (-ve)	PC8 (Dieback)	Propn Stringybark	None	-
Summer Visitors (+ve)	PC3 (Mistletoes) PC10 (% Herb Cover, Herb Height) PC4 (Large Trees) PC2 (Stringybark)	% Herb Cover	None	# Veg. Species # Logs # Box Mistletoe
Summer Visitors (-ve)	PC8 (Dieback)	Dieback	Dieback Dist. Road	Propn Gum-Box Propn Gum Altitude
Winter Visitors (+ve)	PC3 (Mistletoes) PC5 (% Shrub Cover, % Sub-can. Cover, Veg. Biomass) PC9 (Altitude)	# Mistletoe Veg. Biomass	# Veg. Species % Can. Cover Area	Droop. Mistletoe Altitude # Logs Propn Stringybark
Winter Visitors (-ve)	None	Dist. Water-course	# Can. Species	Propn Gum-Box # Veg. Species
Residents (+ve)	None	-	% Acacia	Propn Gum-Box # Logs # Box Mistletoe
Residents (-ve)	PC8 (Dieback)	-	Dieback	Altitude Propn Stringybark Propn Gum

Table 2.7: Summary of the frequency of selection of environmental variables as either positive or negative predictors. The totals are from 68 models generated for the 16 bird groups, using the different statistical analyses. This table is simply a means of viewing the data and should be interpreted with caution. There is a bias towards the variables used to construct the principal components (see Table 2.5 for composition of the principal components). Counts were taken from Table 2.6. Percentages represent the proportion of times each environmental variable was selected by each analysis.

Environmental variable	Total Positive	Total Negative	Total (both ±)	% PCA	% SMR - Spring	% SMR - Seasonal	% HMDS
Dieback	2	31	33	10	9	39	3
Mistletoes (both species)	24	1	25	7	9	6	8
Altitude (PC 9)	13	10	23	4	7	0	16
Propn. Gum-box (PC 2)	7	16	23	0	2	3	12
Propn. Stringybark (PC 2)	14	6	20	6	2	0	12
Area (PC 7)	4	14	18	7	0	13	5
# Logs	16	1	17	0	14	0	11
Distance Edge (PC 7)	3	12	15	7	5	0	3
# Veg. Species (PC 1)	7	8	15	5	5	3	5
Distance Buildings (PC 7)	4	10	14	7	4	3	2
Distance P. Water (PC 7)	2	10	12	7	2	0	1
% Acacia Shrubs (PC 1)	9	3	12	5	3	10	0
% Herb Cover (PC 10)	10	0	10	4	10	0	0
Canopy Height (PC 4)	10	0	10	0	3	0	0
# Shrub Spp. (PC 1)	5	5	10	0	0	0	4
Propn. Gum (PC 2)	2	7	9	0	0	3	7
Tree Diameter (PC 4)	9	0	9	6	0	0	0
Veg. Biomass	9	0	9	2	7	0	2
Tree Density (PC 6)	8	1	9	4	0	0	4
% Can. Cover (PC 6)	7	2	9	4	0	0	0
Large Trees (PC 4)	8	0	8	6	0	0	0
# Sub-canopy Spp. (PC 1)	5	3	8	0	0	3	0
Herb Height (PC 10)	7	0	7	4	0	3	1
% Shrub Cover (PC 5)	4	1	5	2	2	0	1
Dist. Water-course (PC 11)	0	5	5	0	7	0	0
Slope (PC 11)	0	4	4	0	2	4	1
# Cut Stumps	4	0	4	0	3	0	2
% Sub-can. Cover (PC 5)	3	0	3	2	0	0	0
# Dead Trees	2	0	2	0	4	0	0
Propn. Box	1	1	2	0	2	3	0
Isolation	0	1	1	0	2	0	0
Dist. Road	0	1	1	0	0	3	0