

## CHAPTER 1: GENERAL INTRODUCTION

Habitat clearing and fragmentation are primary causes of the decline of bird species in rural areas throughout the world (Robbins 1980; Whitcomb *et al.* 1981; Lynch and Whigham 1984; Opdam *et al.* 1985; Wilcove 1985; Howe 1986; Askins and Philbrick 1987; Freemark 1988; Newmark 1990; Recher and Lim 1990; Saunders and Curry 1990; Robinson 1991; Garnett 1992). Although a reserve system has long been seen as a way to prevent species extinctions in fragmented landscapes, the question of how to select areas for reservation has been the subject of much debate. A helpful start was made by the application of island biogeography theory to land reserves (MacArthur and Wilson 1967; Diamond 1975; Wilson and Willis 1975). This theory proposes that the number of species on an island results from an equilibrium between rates of species immigration and extinction. Consequently species richness will be greatest in larger, less isolated islands.

Many studies of birds in fragmented habitats have indicated that conservation reserves should be as large as possible, suggesting that a conservation priority should be to acquire large reserves (Moore and Hooper 1975; Galli *et al.* 1976; Kitchener *et al.* 1982; Howe 1984; Opdam *et al.* 1985; Askins *et al.* 1987; van Dorp and Opdam 1987; Robbins *et al.* 1989; Newmark 1990). This led to the widely accepted notion that a single large reserve will conserve more species than several small reserves (Diamond 1975; Wilson and Willis 1975). However, a number of authors have demonstrated that there can be as many, if not more bird species in small reserves than large reserves (Higgs and Usher 1980; Loyn and Suckling 1984; MacLellan *et al.* 1986; Verner 1986; Ford 1987, Loman and Von Schantz 1991; McCoy and Mushinski 1994). The resulting debate about whether as many bird species could be conserved in a single large reserve, as in several small reserves became known as the SLOSS debate. In a review of the subject, Simberloff and Abele (1982) found no evidence that larger reserves were better for conserving species richness, than a group of smaller reserves occupying the same total area. The theory of island biogeography is closely associated with the species-area curve, which demonstrates that more species will be present in larger reserves (Connor and McCoy 1979). Simberloff

(1986) argues that, while this aspect of the island biogeography model is supported, few studies support the prediction that rates of immigration and extinction are a function of the size of the reserve. Evidence that the number of species in reserves is the result of an equilibrium between immigration and extinction rates is also lacking.

Soulé and Simberloff (1986) argued that maintaining species richness is not the only, or even the major goal of conservation, and as such the SLOSS debate is no longer an issue in the discussion about the optimum size of reserves. They propose that keystone species whose disappearance would result in a significant decrease in species diversity, be identified. Alternatively, it is argued that indicator species should be identified, the loss of which would herald the loss of many other species (Landers *et al.* 1988). The minimum area required for these keystone or indicator species should then be used as a guide for reserve selection. As a general conclusion, Soulé and Simberloff (1986) stated that reserves should be as large as possible, and there should be many of them.

The growth of minimum viable population analysis marked a shift in the emphasis of conservation biology, from community ecology back to population ecology. The result was an emphasis on the importance of large rather than small reserves for conservation. However, an overriding conclusion from studies of minimum viable populations, is that even the largest existing reserves are too small to conserve many "indicator species" over a long period of time (Davidson 1974; Belovsky 1987; Grumbine 1990; Soulé 1991). This conservation 'crisis' was eased by the metapopulation model, which suggests that a single large reserve is not always necessary for the conservation of indicator or keystone species (den Boer 1981; Gilpin 1987; Opdam 1988; Murphy *et al.* 1990; Howe and Davis 1991). Rather these species may exist as a number of sub-populations that are separate but interacting, and asynchronous in their population fluctuations (Soulé 1987; Simberloff 1988).

It appears that the SLOSS debate has been largely resolved by the absence of sufficient large reserves that are suitable for conserving species. This is particularly true for agricultural landscapes where a network of small reserves is often the only practical option for conserving biodiversity. The metapopulation models indicate that whether conserving a single species or a community in such a landscape, it must be possible for species to move between the several small reserves (or sub-populations). Recently many researchers have stressed the importance of corridors for facilitating this movement in fragmented landscapes (Bennett 1991, 1992; Saunders and de Rebeira 1991; Saunders and Hobbs 1991; Merriam and Saunders 1993). However, in a review of the subject, Simberloff *et al.* (1992) point out that there is little supportive evidence that strips of habitat linking reserves actually serve as thoroughfares. They also argued that as reservoirs for predators and edge species, strips of vegetation may be acting as traps or 'sinks'. Catterall *et al.* (1991) have also questioned the value of strips of vegetation as corridors for bird movement, pointing out that they are often dominated by large aggressive bird species. There are, however, some studies which demonstrate that birds moving through a landscape will use strips of vegetation as fly-ways rather than cross open space (Wegner and Merriam 1979; Finch 1989; Isaacs 1994; Lynch *et al.* in press; for an overview see Bennett 1992). A study by Date *et al.* (1991) also suggests that small patches of vegetation, and even isolated trees, serve as 'stepping stones' for birds moving through disturbed landscapes.

There is a growing realisation that reserve networks will in most cases be insufficient to conserve all species in a landscape (Goldney and Bowie 1990; Pressey 1990; Saunders 1990; Hobbs *et al.* 1992). This emphasises the importance of conservation strategies for off-reserve areas, and the need to put the area of the remnant into a proper perspective relative to other environmental features. There must also be greater flexibility in conservation management. Simberloff and Abele (1982) remarked on the inherent arrogance of proposing that large reserves be set aside with little regard to the social and economic costs of such a conservation strategy.

The importance placed on large remnants for conservation by many authors, is often based on the assumption that beta diversity (diversity within a patch containing several habitats) is of overriding importance compared with alpha diversity (diversity at a given survey point in a single habitat type; see Cody 1975; Haila *et al.* 1994). Some authors have argued that the mechanisms responsible for area effects can more easily be unravelled by measuring alpha diversity, that is, surveying equal portions of habitat in remnants of different size (Simberloff and Abel 1982; Blake and Karr 1987; Freemark and Merriam 1986; Verner 1986). In other words, comparing survey data at a given survey site gives a clearer understanding of the effects of area, than comparing survey data in which a proportionally larger area was surveyed in larger patches. This is the method used in this study and I would like to state clearly at this stage, that throughout this thesis, the relative abundances of the different bird groups and the measures of bird species richness, all refer to counts per site. When bird species richness and abundance is compared in different sized woodland patches, it is counts per site which are being compared, not total counts within the different sized patches.

In previous studies where the same survey effort has been made in different sized remnants, environmental features other than the area of the remnant have been found to be important in predicting the distribution and abundance of bird species (Blake and Karr 1987; Lynch and Wigham 1984). The following is an overview of environmental features, other than the area of the remnant, that have been found to influence bird assemblages. They are often related to, but act independently of the area of the remnant.

It has been known for some time that habitat heterogeneity is one of the mechanisms responsible for area effects (Forman and Godron 1981; James and Wamer 1982; Haila 1983, 1986; Lynch and Wigham 1984; Freemark and Merriam 1986). As larger patches are more heterogeneous, there is a greater diversity of resources available and species diversity increases. One component of habitat heterogeneity that is positively associated with bird species diversity is the number of layers of vegetation. In habitats where

herbaceous vegetation, understorey vegetation and canopy vegetation are all present, there is greater bird species richness (MacArthur 1964; Recher 1969; Karr and Roth 1971; Willson 1974). In a number of Australian studies, understorey vegetation has been found to be positively associated with the number of bird species present (Recher 1969; Loyn and Suckling 1984; Loyn 1985b; Milledge and Recher 1985; Cale 1990; Grover and Slater 1994). Studies in both Australia and overseas, have also found that understorey-dependent species are more susceptible to habitat fragmentation, particularly in grazing landscapes, where grazing pressure increases in smaller remnants (Ford and Bell 1981; Howe *et al.* 1981; Karr 1982; Loyn 1987; Newmark 1990; Grover and Slater 1994). Other studies have shown that understorey vegetation is often more abundant in smaller remnants where there is more edge habitat (Ford 1987; Fuller and Whittington 1987; Fuller *et al.* 1989; Yahner 1993). However, these studies were in forest fragments in the Northern Hemisphere which, unlike the more open eucalypt fragments in Australia, tend to develop shrubby vegetation along the edges. In these Northern Hemisphere forests, migrant bird species appear to be particularly vulnerable to the removal of understorey vegetation (Helle and Fuller 1988; Fuller *et al.* 1989; Hutto 1989).

Bird species richness is not only influenced by vegetation physiognomy or structure, but also by floristics. In a number of Australian studies, the dominant tree or shrub species, or simply the number of vegetation species present, has been found to be a predictor of bird species abundance and or bird species richness (Kitchener *et al.* 1982; Howe 1984; Arnold 1988; Mac Nally 1990b; Leach and Recher 1993; Woinarski and Braithwaite 1993; Wood 1993).

Moderate levels of grazing are likely to increase the extent of herb cover and maintain species diversity in both the herb and understorey vegetation (Duffy 1974; Miller and Watson 1974; Connell 1979). Under such a grazing regime, it is likely that as many bird species are favoured as are disadvantaged (Robbins *et al.* 1989; Saunders and Curry

1990). However, where there is intensive prolonged grazing the understorey and herbaceous vegetation is removed, resulting in less structural heterogeneity, fewer vegetation species, and a subsequent loss of bird species (Veblen *et al.* 1991; Bock *et al.* 1992; Scougall *et al.* 1993; Fleischn er 1994). The bird species that are worst affected by excessive grazing in Australian ecosystems are understorey-dependent species and ground-foragers (Howe *et al.* 1981; Recher and Lim 1990; Garnett 1992; Grover and Slater 1994).

Another aspect of vegetation structure is tree age. As trees get older they provide more resources for bird species, such as nesting hollows (Mackowski 1984), increased amounts of loose bark and canopy foliage, and increased numbers of mistletoes (Loyn 1985b). The presence of large trees has been found to influence bird assemblages in Australia (Kavanagh *et al.* 1985; Saunders *et al.* 1985; Saunders and Ingram 1987; Braithwaite *et al.* 1989) and elsewhere (James and Wamer 1982; Lynch and Wigham 1984; Mannan and Meslow 1984; Fuller *et al.* 1989; Hutto 1989; Tellaria *et al.* 1992). The overseas studies have also found that bird species richness is greatest where intermediate-aged stands occur (Fuller and Moreton 1987; Fuller *et al.* 1989; Fuller and Steel 1990; Fuller and Henderson 1992). Bird species that depend on large trees are particularly susceptible to fragmentation in rural Australia because larger trees are more likely to be harvested in the smaller, more accessible remnants (Robinson 1991; see also Garnett 1992). There also appears to be little replacement of trees in smaller remnants where heavy grazing prevents regeneration (Bennett *et al.* 1994). The presence of fallen timber also influences bird assemblages in both Australian and overseas habitats (Davidson and Davidson 1992; Smith *et al.* 1992; Haila *et al.* 1994). Tubbs (1974), Bennett (1992) and Recher (1993) have all recommended that logs should be retained in remnant vegetation as a means of restoring habitat quality and increasing bird species richness.

An increase in the proportion of edge habitat in smaller remnants is one of the likely causes of area effects on bird assemblages. Edge habitat, being more exposed, is subject to increased radiation and water fluxes that result in changes to the microclimate and vegetation structure. Increased wind along remnant edges also removes leaf-litter and loose bark from trees (Laurance and Yensen 1991; Saunders *et al.* 1991). Changes in the bird community along habitat edges have been described in Australian studies (Loyn 1985a, Catterall *et al.* 1991; Lynch and Saunders 1991). However, given the open nature of many eucalypt habitats, edge effects appear to be less obvious than those frequently recorded in Northern Hemisphere studies (Johnston 1970; Gates and Gysel 1978; Morgan and Gates 1982; Gotryd and Hansel 1986; Rosenberg and Raphael 1986; Fuller and Whittington 1987; Temp e and Cary 1988).

Landscape features, other than the area of a remnant, also influence bird assemblages. Isolation effects have been described in some Australian studies (Ford and Howe 1980; Howe *et al.* 1981; Howe 1986; Cele 1993), but not in others (Kitchener *et al.* 1982; Howe 1984). Where isolation effects do occur they are consistent with predictions based on island biogeography theory. Generally, in both Australia and overseas, there is a decrease in the number of bird species as the amount of habitat surrounding a remnant decreases and the remnant becomes 'isolated' (Karr 1982; Lynch and Wigham 1984; Askins *et al.* 1987; Ford 1987; van Dorp and Opdam 1987; Freemark 1988; Robbins *et al.* 1989). Andrén (1994) and Green (1994) have predicted that the habitat surrounding a remnant can be cleared to a critical point, beyond which both the isolation and area of the remnant will begin to influence the bird assemblages within the remnant. Altitude is another landscape feature that appears to influence bird species richness. The changes in the bird community at different altitudes appear to be related to the residency status of the bird species. Changes in nutrient availability, vegetation type and structure, and climatic changes at different altitudes also appear to influence bird assemblages (Braithwaite *et al.* 1989; Fuller and Crick 1992; Brown and Stillman 1993; Janes 1994). The proximity of water and water courses are other landscape features that have been

found to influence the distribution of bird species in Australian landscapes (Howe *et al.* 1981; Loyn 1985b; Smith 1985; Saunders and Curry 1990; Grover and Slater 1994).

The aforementioned studies demonstrate that bird species assemblages within a habitat remnant can be influenced by a range of environmental features, that may or may not be related to the area of the remnant. Other studies have demonstrated that the response of bird species to area may also be due to interactions with other bird species. In many Australian landscapes, large territorial species such as the noisy miner *Manorina melanocephala* are more common in smaller woodlots, and exclude most other bird species from their territory (Howe *et al.* 1981; Loyn 1985a; Catterall *et al.* 1991; Grover and Slater 1994). Increased nest predation has been suggested as a reason for the decline of smaller birds in small woodland remnants on the New England Tablelands in northeast New South Wales, Australia (Ford and Bell 1981, Barrett *et al.* in prep.). Also in northeast New South Wales, Howe (1986) found that nest predators were more common in fragmented landscapes. A similar pattern was described by Saunders and Curry (1990) in the Western Australian wheatbelt. Both nest predation and nest parasitism have also been implicated in the decline of small songbirds, particularly neotropical migrants, in fragmented landscapes in the Northern Hemisphere (Robbins 1980; Ambuel and Temple 1983; Wilcove 1985; Askins and Philbrick 1987; Temple and Cary 1988; Yahner and Scott 1988; Terborgh 1989; Telleria and Santos 1992).

Throughout Australia many bird species are declining in range and becoming locally extinct (Howe 1986; Saunders 1985; Goldney and Bowie 1990; Woinarski and Braithwaite 1990; Robinson 1991; Garnett 1992; Paton *et al.* 1994). Clearance and fragmentation of habitat, grazing (particularly by sheep) and predation are the primary causes of this decline in bird species (Recher and Lim 1990; Garnett 1992). Local extinctions include most types of birds. However, it is the ground-foragers, ground-nesters and species requiring large nesting-hollows that are particularly vulnerable to extinction. While the causes of species decline in Australia are widespread and complex, there seems little doubt that, with a



commitment to careful management, these trends can be halted and even reversed (Saunders *et al.* 1987; Bennett 1992; Hobbs *et al.* 1993; Recher 1993).

The present study is concerned with bird-habitat relationships and the conservation status of birds on the New England Tablelands, in northeast New South Wales, Australia. The study region is the Armidale Plateau, an upland area within a 60 km radius of Armidale. The original state of the avifauna on the New England Tablelands is not well documented. However, the available evidence suggests that widespread changes to the landscape have had a marked impact on regional bird species assemblages (Ford and Bell 1981; Howe 1984, 1986; Ford *et al.* 1985; Dunkerley 1989; Ford and McFarland 1991; Ford *et al.* 1995). Diaries from the first explorers, and early photographs of the region, indicate that woodland cover extended throughout most of the Tablelands (Curtis *et al.* in press). Today about half of the New England Tablelands area is timbered (Smith and Turvey 1977), and on the Armidale Plateau this figure is closer to 20%. As such, it is likely that declines in bird species abundance occurred as a result of habitat clearing and fragmentation.

It also appears that continued grazing by sheep and cattle, a practice which has occurred since the early days of settlement, has reduced understorey vegetation in a large proportion of the woodland remnants on the Armidale Plateau (Ford and Bell 1981; Morgan 1981). Eucalypt dieback threatens to wipe out much of the remaining woodland cover over large areas of the Tablelands. The dieback phenomenon is thought to have been caused by the excessive clearing of woodland which, in combination with pasture improvement and intensified stocking rates that have been practiced since the 1940's, have led to outbreaks of leaf-eating insects (Lowman and Heatwole 1992). There appears to be a negative feedback loop operating in woodland sites where dieback is extensive. While dieback is likely to cause a decline in bird numbers as a result of defoliation (Ford and Bell 1981), experiments indicate that the exclusion of birds from eucalypt branches results in further dieback because fewer leaf-eating insects are being removed by birds (Dunkerley 1989; Ford 1989). The role of birds as a biological control for insects has been noted by other authors (Dow

1977; Holmes *et al.* 1979; Loyn *et al.* 1983; Holmes and Recher 1986; Landsberg *et al.* 1990; Davidson and Davidson 1992).

Predictive models can be constructed to generate hypotheses about the habitat requirements of birds (Temple and Wilcox 1986). Until such hypotheses can be tested experimentally, it is legitimate to use these predictive models as guidelines for management in landscapes where bird species are declining. Chapter 2 of this thesis is a presentation of predictive models, based on a survey of birds and a description of the environment, in 294 woodland sites on the Armidale Plateau. Bird species were combined into groups based on taxonomy, feeding guild and habitat preferences. Because the aim of this study was to identify the environmental features that were most likely to be influencing each bird group, species interactions, such as the presence of nest predators or noisy miners, were not built into the predictive models (as suggested by Laymon and Barrett 1986). The models were constructed using different statistical and survey methods, allowing an assessment of the consistency of the models. Environmental features that appear to influence bird species richness were identified and discussed in relation to land management practices on the Armidale Plateau.

The third chapter investigates temporal changes in the bird community in relation to the size of the woodland patch, the dominant tree species present and the presence of noisy miners. The sites used for this latter analysis were a subset of those used for building the predictive models in Chapter 2. The third chapter is also concerned with the abundance of bird species in grassland relative to strips of woodland and woodland patches. The value of strips of woodland and water-courses as fly-ways was assessed by counting the frequency and direction of long-distance flights (> 40 m) in strips of woodland, along water-courses, and also in patches of woodland and on grassland.

The fourth chapter is an assessment of the conservation status of 137 land birds that occur on the New England Tablelands. This assessment is largely based on the previous two

chapters, however, new data describing the response of individual bird species to size of the woodland patch are presented. The birds are divided into five groups based on their relative abundance and their response to habitat disturbance. A conservation strategy is proposed for the avifauna of the region. Chapter 5, the final chapter, is a general discussion which provides an overview of the results and conclusions. This chapter also provides further comment on some of the risks associated with applying ecological theory from the Northern Hemisphere to Australian ecosystems.