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Vertebrate Biodiversity on Australian Rice Farms

An Inventory of Species, Variation Among Farms, and Proximate Factors Explaining that Variation

A report for the Rural Industries
Research and Development
Corporation

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Foreword

An increasing body of evidence indicates that considerable biodiversity can be maintained in ecosystems that have been modified for human use. One such system may be the Australian rice agroecosystem of the Riverina Bioregion, New South Wales. Although the vertebrate diversity of the Riverina Bioregion is relatively well known, the occurrence of this fauna occupying rice farms in that region is has not been quantified. A baseline survey of vertebrates, and research into what underpins that diversity is necessary in order to formulate management strategies (on and off farms) to maintain or enhance that diversity. As part of an environmental initiative of the Australian rice-growing industry (RGA), we surveyed vertebrate biodiversity for three field seasons (2000-2003).

This publication details our findings from an (1) extensive study of vertebrate biodiversity on 10 rice farms in three irrigation areas, and from two intensive studies: (2) a study of the influence of on-farm vegetation remnants on bird diversity, and (3) a study of how turtles use rice farms. It integrates knowledge gained in these studies with that of other studies to formulate management strategies for rice farmers and other stakeholders in the region.

This project was funded from industry revenue, which is matched by funds provided by the Australian Government.

This report, an addition to RIRDC's diverse range of over 1500 research publications, forms part of our Rice R&D program, which aims to improve the profitability and sustainability of the Australian rice industry.

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Peter O'Brien
Managing Director
Rural Industries Research and Development Corporation

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Executive Summary

An increasing body of evidence indicates that considerable biodiversity can be maintained in ecosystems that have been modified for human use. One such system may be the rice agroecosystem of the Riverina Bioregion, New South Wales, Australia. Although the vertebrate diversity of the Riverina Bioregion is relatively well-known, the occurrence of this fauna occupying rice farms in that region is has not been quantified. A baseline survey of vertebrates, and research into what underpins that diversity is necessary in order to formulate management strategies (on and off farms) to maintain or enhance that diversity. As part of an environmental initiative of the Australian rice-growing industry (RGA), we surveyed for vertebrates on 10 rice farms in three irrigation areas in the Riverina for three field seasons (2000-2003). We used a variety of field techniques to document an inventory of 201 species of vertebrates on the farms, including 149 bird species, 23 reptiles, 19 mammals, 7 frogs, and 3 fishes. Six threatened species were identified, including five bird species and one frog species, and we found evidence that at least two of these (the southern bell frog and the superb parrot) were breeding on the farms. Thus, our findings (1) support other studies maintaining that considerable vertebrate diversity occurs in the Riverina; (2) extend those findings to the rice farm scale; and (3) demonstrate that the Australian rice agroecosystem, like most ecosystems, contains an assemblage of species (e.g., threatened species) in need of monitoring and/or management.

Two major factors underpinning existing biodiversity on rice farms were: productive rice bays and remnant vegetation patches. Rice bays were inhabited by large numbers of frogs, snakes, turtles, and waterbirds. We estimated that hundreds of millions of frogs of one common species, the spotted grass frog, are produced on rice farms annually, based on pitfall captures of dispersing frogs. This species probably represents the highest biomass of any vertebrate on rice farms. The importance of frogs on rice farms is further highlighted by our anecdotal finding that brownsnakes consumed frogs on rice farms, despite their documented habit of consuming mostly lizards and small mammals. We found a rapid increase in the average richness and abundance of birds on farms after the flooding of rice bays (11+ species, 140 individuals) compared to pre-flooding (7 species, 40 individuals), and a similar decrease in bird richness and abundance was associated with draining of the rice bays. Based on capture rates and bird counts in ‘wet’ and ‘dry’ habitats within farms, rice farms are likely to have a higher abundance of frogs, snakes, turtles, and birds, and higher species richness of birds, than dryland farms. Farms with remnant vegetation patches were occupied by more lizard species (on average, 6 vs. 2) and woodland specialist birds (on average, 7 vs. 2) than were farms without remnants. Lizards were also more abundant on farms with vegetation remnants. Farms with remnant vegetation patches are likely to have higher species richness of birds, frogs, lizards, snakes, and mammals (mostly ferals), and higher abundance of birds, than farms without remnant vegetation patches.

The apparent importance of vegetation remnants on birds in the general survey prompted us to study the relationships between characteristics of remnants (e.g., size, shape, condition) and bird diversity. An intensive study of 23 Black Box remnants in the Coleambally Region during 2002 indicated that (1) patch area and shape of remnants was positively related to DECREASER bird and WOODLAND bird richness, but inversely related to INCREASER bird abundance (see text for explanations of terms); (2) habitat complexity, or the condition of remnants was positively related to WOODLAND bird richness and abundance; (3) some woodland bird species known to be in decline in the region were found only in large remnants with a high area-to-perimeter ratio in relatively good condition (i.e., some understory); (4) smaller remnants generally contained bird species known to be common in the region; and (5) noisy minor abundance did not appear to influence species richness or abundance of any other bird groups.

An intensive study of the habitat utilisation, movements, and diet of longneck turtles revealed that (1) turtles exploited the seasonally-available flooded rice bays, using them as a surrogate for natural wetlands; (2) however, invertebrate prey productivity was apparently not higher in rice bays than in irrigation channels; (3) turtles used all major aquatic habitats on rice farms; (4) some turtles moved long distances between farms; (5) turtles became more selective of prey types as prey richness increased; (6) timing of movements between habitats differed between males and females; and (7) mortality and injury occurred when turtles attempted to move through dethridge wheels.

Our study provided evidence in support of management recommendations provided by others, and these recommendations can be amalgamated into two major on-farm recommendations: (1) improving existing vegetation; and (2) revegetation. Vegetation remnants and riparian vegetation are among the rarer habitats in and around rice farms in the Riverina. Accordingly, fauna associated with these habitats are less common. Other recommendations include (3) conserving existing vegetation, in the context of any expansion of the industry; (4) maintaining biodiversity in rice bays. Finally, other studies support some additional recommendations that we did not address in the present studies, but need to be integrated. These include (5) avoiding over-grazing of areas with remnant vegetation; (6) avoid removing fallen timber, bark, rocks, and leaves from areas with remnant vegetation; and (7) improving and revegetating riparian areas.

We suggest that the best conservation strategy to protect, maintain, or enhance biodiversity on rice farms should include the following three approaches: First, a monitoring program should be developed and initiated, whereby populations of selected species are sampled every few years. Second, an experimental approach improving rare habitats (e.g., planting or enhancing an understory in vegetation remnants) would provide a low-risk conservation measure for key species. Third, ecological research into selected species (e.g., species occupying rarer habitats) would provide critical information on habitat utilisation on farms, thereby facilitating their monitoring and management. The latter should include threatened species to uphold the goal of maintaining current levels of biodiversity.

Specific recommendations stemming from the ‘birds and remnants’ study include (1) maintaining, setting aside, or revegetating remnant vegetation patches > 10 ha (however, patches > 50 ha are best, and any patch is better than none); and (2) improving habitat complexity (e.g., understory structure) in remnant vegetation patches. Specific recommendations from the ‘turtles in rice’ study include (1) general revegetation or plantings as refuges for turtles traveling overland; (2) avoiding the use of fences that form barriers to overland movement of turtles; and (3) research to quantify the impact of dethridge wheels on mortality and injury of turtles.

Introduction

An important recent advance in conservation biology is the realisation that we cannot hope to conserve the earth's species through protected reserves alone (Soule and Sanjayan, 1998; Rosenweiz, 2003). At least 95% of the earth's land surfaces have been "refashioned" by humans (Srivasta *et al*., 1996; Rosenwiez, 2003), and these areas are likely to be subject to further modification. Increasingly isolated habitat patches, in reserves and otherwise, are earmarked for increases in extinction rates through natural (e.g., drought, fire) and unnatural (e.g., greenhouse events) processes (Rosenwiez, 2003). How can society reduce current and forecasted high extinction rates?

Given the unlikely prospect of restoring ecosystems or habitats and their biodiversity to their original conditions, we are faced with the problem of reconciling biodiversity conservation with human land use. The philosophy, science, and practice of conservation must be framed against the reality of human-dominated ecosystems, rather than the separation of humanity and nature underlying the modern (traditional) conservation movement (Western and Pearl, 1989; Srivasta *et al*., 1996; Knight, 1999; Western, 2001). For example, traditionally biodiversity conservation has been thought to be best practiced through the establishment of reserves (Margules and Pressey, 2000; Rodrigues and Gaston, 2001), but this approach may have limitations in relatively heterogeneous production landscapes outside protected areas (Fischer *et al*., 2004). Yet, production areas dominate the earth's surface and are increasing (Srivasta *et al*., 1996).

The strategic inclusion of a new branch of ecology termed 'reconciliation ecology', or the 'maintenance of species friendly habitats in the places where humans live, work, and play', may be our only chance of conserving biodiversity (Rosenweiz, 2003). Reconciliation ecology seeks environmentally-friendly ways for us to continue to use the land for our benefit. Reconciliation ecology has already enjoyed some success as a conservation strategy (see examples in Rosenweiz, 2003). A major driver in reconciliation ecology is likely to be restoration ecology (see Saunders *et al*., 1993; Dobson *et al*., 1997). Restoring or improving habitats for biodiversity, however, requires detailed knowledge of biodiversity and what factors underpin it.

An increasing body of evidence indicates that considerable biodiversity can be maintained in ecosystems that have been modified for human use (Knight, 1999; Rosenweiz, 2003; but see Lemly *et al*., 2000). One such system is the irrigated landscape. For example, although wildlife use of farms has received very limited study in general (Lindenmeyer *et al*., 2003), irrigated rice agroecosystems have been shown to harbour considerable waterbird diversity (Fasola, 1986; Heitmeyer *et al*., 1989; Miller *et al*., 1989; Remsen *et al*., 1991; Pain, 1994; Brouder and Hill, 1995; Fasola *et al*., 1996; Elphick and Oring, 1998; Lane and Fujioka, 1998; Elphick and Oring, 2003; Maeda, 2001; Elphick, 2004), and frogs (Maeda and Matsui, 1989; Fujioka and Lane, 1997), and in some cases rice fields have been suggested to be 'functionally equivalent' to natural or semi-natural wetlands for some fauna (Fasola and Ruiz, 1996; Fujioka and Lane, 1997; Elphick, 2000). On the other hand, irrigated systems have resulted in the degradation of wetlands and their biodiversity around the world (Gerakis and Kalburtji, 1998; reviewed in Lemly *et al*., 2000) and the general trend across agricultural landscapes is that, as farming intensity increases, biodiversity decreases (Wood *et al*., 2000; Donald *et al*., 2001). In either case, understanding and maintaining biodiversity in rice agroecosystems is of considerable importance, because rice occupies a larger area than any other crop in the world (Fores and Comin, 1992), and about 40 % of the world's population depends on rice for food (Odum, 1993). This situation will likely be even more critical in the future because the world's human population is expected to at least double before it stablises, and the need for primary products is expected to triple in the next 50 years (Avery, 1996).

The Australian rice agroecosystem is essentially restricted to the Riverina Bioregion, which encompasses over 10,000,000 ha in southern New South Wales and northern Victoria (Benson, 1999). About 150,000 ha of rice is produced annually in this region, and rice farmers typically grow a mixture of rice and other (dry) grains on small (e.g., 400 ha) parcels of land in an irrigated landscape. Although the general vertebrate diversity of the Riverina Bioregion is relatively well-known on a large scale (through field guides, etc.), the occurrence of this fauna on rice farms in that region has not been quantified. A baseline inventory of vertebrates, and research into what underpins that diversity is necessary in order to formulate management strategies (on and off farms) to maintain or enhance that biodiversity.

As part of an environmental initiative of the Australian rice-growing industry (RiceGrowers' Association), we launched a three-year, three-part study of vertebrates inhabiting rice farms in the Riverina Bioregion. The study included an extensive survey of vertebrates occupying rice farms (Doody, 2002; Doody and Osborne, 2003; Doody et al., 2004), and two intensive studies, one attempting to determine what features of vegetation remnant patches are important for birds (Bourne, 2003), and another aiming to determine how turtles utilise rice farms (Rennie, 2002).

In the extensive study, we surveyed for vertebrates on 10 farms broadly located over three irrigation areas in the rice-growing Riverina for three field seasons (2000-2002). In this study our major objectives were threefold: (1) determine what vertebrate wildlife species comprise on-farm biodiversity, (2) to determine what features of rice farms likely underpin that biodiversity, and (3) provide recommendations that allow farmers to make informed decisions regarding the co-management of on-farm biodiversity and crop production. We hypothesised *a priori* that vertebrate biodiversity would be positively influenced by the presence of on-farm remnant vegetation and the distance from a major river corridor.

The finding by us and others that bird diversity was associated with the presence of vegetation remnants prompted us to intensively study how bird abundance and species richness varied with remnant vegetation patch characteristics in a highly modified agroecosystem.

In southern regions of Australia, woodland communities have been severely affected by the clearance of native vegetation for agriculture (State of the Environment Advisory Council 1996; Er *et al.*1998; Ford *et al.,* 2001; Major *et al.* 2001). In areas such as the wheatbelt of Western Australia (Saunders, 1989) and the agricultural regions of western New South Wales (Benson, 1999) habitat clearance is especially widespread, with tree cover commonly below 10%. The conservation of these habitats is crucially important, as woodland communities are the habitat of preference for 149 species of birds (Recher 1985). Not surprisingly, these habitats contain the highest concentration of threatened bird species in Australia (Garnett, 1992; Robinson and Trail, 1996; Major *et al.*, 2001). The remnants of woodland remaining in the landscape are often severely fragmented and degraded (Major *et al.*, 2001), and are subjected to the continued declines of woodland bird species even after the initial habitat loss as a result of degrading processes such as overgrazing, tree dieback, changed fire regimes and interspecific competition of invasive species (Saunders, 1989; Barrett *et al.,* 1994; Robinson and Trail, 1996; Recher, 1999; Reid, 1999; Ford *et al.,* 2001). Habitat fragmentation has been shown to influence the composition of bird assemblages, and bird species richness and abundance (e.g., Major *et al.*, 2001; Seddon *et al.*, 2001). Few studies have sought to examine how factors at the landscape and patch scales influence the occurrence of birds and bird communities (Graham and Blake, 2001; Major *et al.*, 2001). The Riverina bioregion of central southern NSW presents a good example of a highly modified agricultural matrix landscape in which to study how remnant vegetation patches influence bird species richness and abundance.

We studied bird diversity in remnant vegetation patches on 23 farms that varied in size, shape, and condition (complexity) during spring and summer 2002. We hypothesised that bird species richness and abundance would increase with increasing patch area and structural complexity, but would decrease or remain unchanged with increasing edges. The amount of habitat needed for a particular group or species is currently poorly understood (Lindenmeyer *et al*., 2003). In addition, because the noisy miner (*Manorina melanocephala*) has been shown to decrease species richness and abundance of small passerines, most notably insectivorous and nectarivorous species, in small, degraded remnants (Loyn, 1987; Grey *et al*.,1997, 1998; MacNally *et al*., 2000; Ford *et al*., 2001; Major *et al*., 2001), we tested this hypothesis on the study farms.

Finally, our early finding of large numbers of long-necked turtles (*Chelodina longicollis*) on one of the extensive study farms prompted us to determine how this species utilises rice farms. Such a study is appropriate because (1) managing wildlife requires an understanding of how animals utilise their habitats (e.g., Recher *et al.*, 1987); and (2) there are limitations associated relying on species richness data to achieve conservation objectives (Hazell *et al*., 2001) .

The eastern long-necked turtle, *Chelodina longicollis*, a native freshwater species, is common and widespread in eastern Australia (Cann, 1998). The species is known to inhabit a wide range of water bodies including permanent lakes, rivers, and ephemeral habitats such as swamps, wetlands and billabongs, and farm dams (Parmenter, 1976; Chessman, 1978; Stott, 1987; Chessman, 1988; Kennett and Georges, 1990). In particular, *C. longicollis* has a propensity to exploit ephemeral habitats such as wetlands and swamps (Chessman, 1978; Kennett and Georges, 1990), presumably because its invertebrate prey reach high densities in these habitats, particularly after a complete cycle of drying and refilling (Swanson and Meyer, 1977; Brinson *et al.*, 1981; Dannell and Sjoberg, 1982). For example, *C. longicollis* near Jervis Bay, NSW resided in permanent lakes during dry periods, where they experienced little or no growth and reproduction (Kennett and Georges, 1990). However, after a nearby ephemeral swamp filled following heavy rainfall, the turtles moved into the swamp to exploit high invertebrate densities, thereby promoting growth and reproduction in the population.

We hypothesised that *C. longicollis* might display a similar movement pattern in the rice agroecosystem. This hypothesis was based on (1) sightings by farmers of long-necked turtles in and around rice bays; (2) our preliminary captures of long-necked turtles in irrigation channels; and (3) knowledge of the hydrological cycle on rice farms. Typically, an Australian rice farm consists of semi-permanent to permanent irrigation channels and seasonally irrigated rice bays that generally hold water from September/October to February/March. In theory, the turtles could spend most of their time in the more permanent irrigation channels, but then exploit the high productivity of invertebrate prey in the seasonally-available, flooded rice bays. We tested this hypothesis by intensively studying habitat use and movement patterns of *C. longicollis* on a typical rice farm near Finley, New South Wales. For this, we radio-tracked 14 turtles (7 females, 5 males) from September 2001 to March 2002. To test a prediction of the hypothesis, that rice bays are rich in prey productivity relative to the irrigation channels, we quantified prey availability (richness and abundance) in the two habitats, and the prey consumed by the turtles (stomach contents). We also examined differences between the sexes in habitat use, movements, and prey preference.

Objectives

The Extensive study

(1) determine what vertebrate wildlife species comprise on-farm biodiversity;
(2) determine what features of rice farms likely underpin that biodiversity; and
(3) provide farmers with management recommendations to allow informed decisions regarding the co-management of on-farm biodiversity and farm production

The Intensive Studies

I. The Influence of Size, Shape, and Habitat Condition of Remnant Vegetation Patches (Black Box) on Bird diversity
  (1) determine the relationship between remnant vegetation patch characteristics (size, shape, condition) and bird species richness
  (2) determine the relationship between remnant vegetation patch characteristics (size, shape, condition) and bird species abundance
  (3) elucidate patch and landscape requirements for bird species
  (4) to provide the above findings to land managers to better equip them to manage on-farm bird diversity

II. Habitat Utilisation, Movements, and Diet of Long-necked Turtles
  (1) determine habitat use of *C. longicollis* on a rice farm
  (2) determine movement patterns of *C. longicollis* on a rice farm
  (3) quantify the diet of *C. longicollis* on a rice farm
  (4) provide information on habitat requirements of *C. longicollis* to farmers to allow informed decisions regarding how management of turtles using rice farms

Methodology

I. The Extensive Study

An Australiasian grebe (*Tachybaptus novaehollandiae*)

A. The Inventory

Study Area, Study Farms, Study Period, and Study Species

The Riverina Bioregion

The Riverina is located in central southern New South Wales (NSW) (Fig. 1) and is classified as a bioregion within the Interim Biogeographic Regionalisation for Australia (IBRA) (Thackway and Creswell, 1995). Two-thirds of the bioregion is situated in NSW, while the remainder is situated in Victoria (VIC) (Benson, 1999). The Riverina Bioregion covers 6,891,966 hectares of NSW and has been cleared of 31% of its woody vegetation (Benson 1999). This figure is somewhat misleading, as within the bioregion 76% of the Murray Province in the central division has been cleared wile only 4% of the Lachlan Province in the western division has been cleared (Eardley, 1999). The average clearance figure also fails to take into account the clearance of non-woody vegetation such as chenopod shrublands that were once widespread and have now been replaced by native grasslands (Benson, 1999; Eardley, 1999).

The Riverina is the most poorly protected bioregion within NSW, having only 0.4% of its total area contained within conservation reserves (Benson, 1999). However, other bioregions within NSW such as the Darling Riverine Plains (0.9%), Mulga Lands (1.1%) and the NSW South-western Slopes (1.2%) are also heavily under-represented in conservation reserves (Benson, 1999).

The geology of the area is dominated by fluvial plains composed of alluvial and lacustrine deposits of gravel, sand, silt and clay (Cunningham et al., 1981; Porteners, 1993). These deposits were initially formed in the late Tertiary Period and are still being built upon today (Porteners, 1993). The soils of the bioregion are largely composed of red clay or grey cracking clay (Benson, 1999).

The Riverina Bioregion has a dry climate, receiving its precipitation at a relatively constant rate throughout the year. It is characterised by hot summers and cold winters. The vast majority of eastern Australia was subjected to an intensive drought in 2002 (Bureau of Meteorology, 2003). This was also strongly evident in the local area with reduced precipitation (lowest on record in Coleambally) and increased maximum temperatures for most of the year.

The major land uses in the Riverina are cropping and grazing. Many properties are subject to irrigation to boost production of crops such as rice, winter cereal crops, citrus, and provide better grazing.

Study Area - Development of the Australian Rice-growing Industry

Rice production in Australia occurs predominately in the Riverina Bioregion of New South Wales. About 2,500 family-operated farms grow rice within three major irrigation districts: the Murrimbidgee, Coleambally, and Murray Valley (Fig.1). The first irrigation farms were established along the Murray River in the 1880’s (Davidson, 1988; Eastburn, 1990). Irrigation systems continued to develop extensively throughout the Murray Region during the 1900’s (Davidson, 1988). Today, there are more than 6000 km of irrigation channels in the Murray-Darling Basin (Bowmer et al., 1994). The development of irrigation further enabled the production of rice crops, and by the late 1920’s Australia had become self sufficient in the commercial production of rice (Dargin, 1941). The regional economy is now dependent on water diverted from the Murray and Murrimbidgee rivers, which supplies water to produce some 50% of Australia's rice crop (Zanatta et al., 2001). Rice is now the predominant summer crop in the NSW Riverina, with approximately 120,000 hectares of rice sown annually within and around this irrigation district (Bowmer et al., 1998).

The development of much of the NSW rice-growing district has been characterised by loss of natural habitat, and the impact on biodiversity appears to have been substantial (Mullins and Sutherland, 2001). During early agricultural development in the 1920’s, the Murray Valley was cleared extensively for grazing and crop production (Glanznig, 1995), contributing to the current restriction of woody vegetation to only 24% of the landscape (Eardley, 1999).

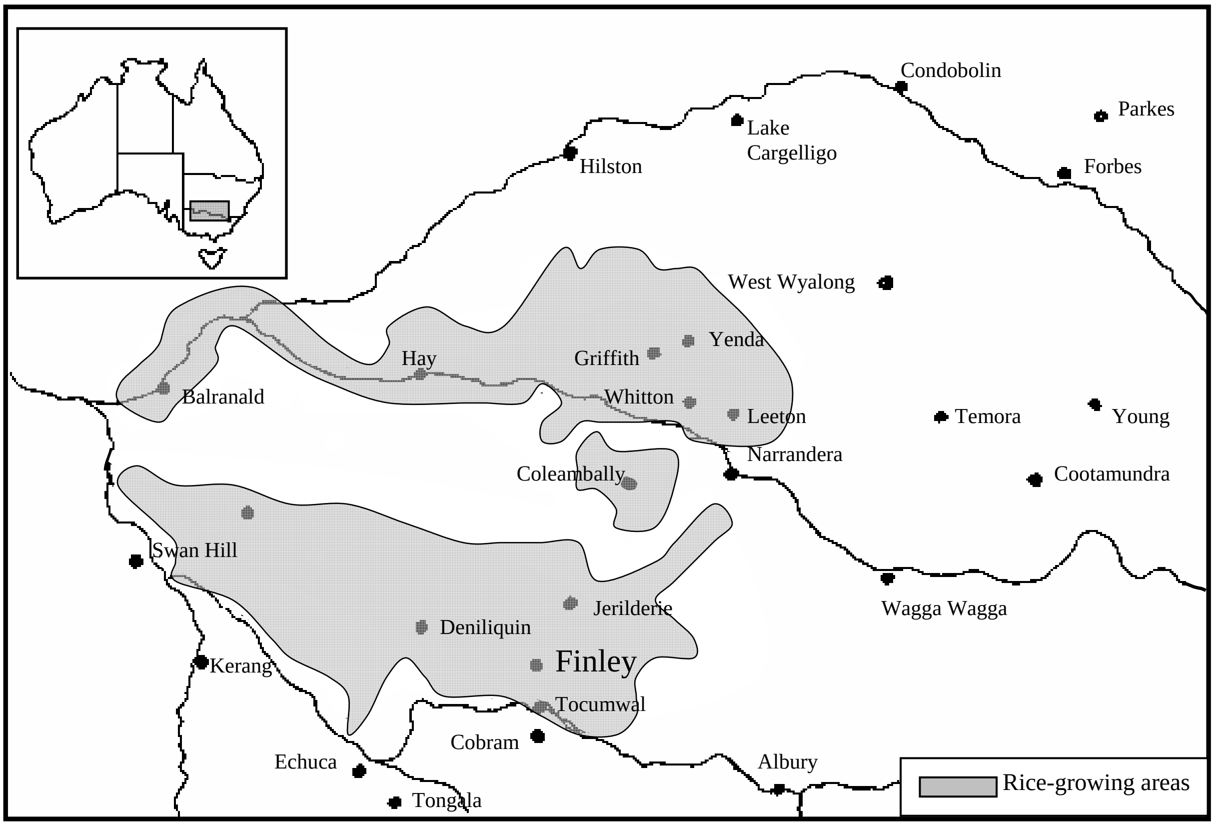


Figure 1. Map of the rice-growing region within the NSW Riverina showing the three main irrigation areas: Murrimbidgee (top), Coleambally (middle), and Murray Valley (bottom). Insert shows location of the Riverina Bioregion in Australia.

Study Farms

The extensive study included 10 rice farms: four in the Murrimbidgee Irrigation Area (MIA), two in the Coleambally Irrigation Area (CIA), and four in the Murray Valley (MV) (Table 1). These farms were chosen based on three criteria: (1) the presence of on-farm remnant vegetation; (2) the distance to a major river corridor; and (3) the willingness of the farmer to facilitate our surveys. To test the hypothesis that vertebrate biodiversity is positively influenced by the presence of remnant vegetation and the distance to a major river corridor, we chose farms such that some contained remnants and some did not, and some were close to river corridors and some were not.

One of the farms (Old Coree), was designated a ‘pseudocontrol’ site, or a site that might best approach pre-rice farming levels of vertebrate biodiversity. This property encompasses a large (~ 900 ha) vegetation remnant and a riparian corridor (Billabong Creek), and would provide a benchmark for achieving maximum levels of biodiversity. However, the site is somewhat atypical in possessing sandhill habitat. Thus, comparisons between this site and other farms should be made with caution (hence the term ‘pseudocontrol’ site).

Table 1. Habitats into which pitfall trap-lines were employed on 10 farms in three irrigation districts during 2000-2002 (the extensive study). When two habitats are given trap-lines were placed directly in the ecotones with one side of each fence facing each habitat. Dry crops = canola, wheat, or fallow. S = sandhill, cs = chenopod shrubland, bb= black box depression, rg = river red gum forest, r = revegetated plot. MV = Murray Valley Irrigation Area; CIA = Coleambally Irrigation Area; MIA = Murrimbidgee Irrigation Area.

fence #	farm	irrigation district	habitat type
1	Old Coree	(MV)	remnant (s/cs)
2	Old Coree	(MV)	rice/remnant (bb)
3	Marsden	MV	rice/dry crop (dam)
4	Marsden	MV	dry crop/dry crop
5	Stephenson	MV	rice/dam
6	Stephenson	MV	remnant (bb)
7	Hall	MV	rice/dam
8	Ewan	CIA	remnant (bb)/rice (dam)
9	Ewan	CIA	rice/rice
10	Hogan	CIA	dry crop/dry crop
11	Hogan	CIA	rice/remnant (bb)
12	King	(MIA)	remnant (rg)
13	Woodside	MIA	rice/remnant (bb)
14	Woodside	MIA	remnant (bb)
15	Houghton	MIA	remnant (bb)
16	Houghton	MIA	rice/remnant (r)
17	Draper	MIA	rice/disturbed
18	Draper	MIA	dry crop/dry crop

Study Period

The extensive study spanned three field seasons (2000-2002). To maximise the probability of detection, especially of ectotherms (reptiles, frogs), surveys were conducted during the warmer months (September-April) each year.

Study Species

In the present study we surveyed for vertebrates (birds, reptiles, mammals, amphibians). Fish were not targeted but were noted opportunistically. Bats were only targeted for a short period during 2002, in response to low capture or detection rates of other groups/species associated with a severe drought.

Field Techniques – I. Pitfall trapping

Pitfall Trapping as a Detection Technique

Pitfall trapping is a widely-used method to capture small animals (Heyer et al., 1994), and is proven effective in capturing Australian reptiles, mammals, and frogs (Braithewaite, 1983; Friend et al., 1989; Hone et al., 1992; Downey and Dickman, 1993; Predavec and Dickman, 1993). Pitfall traps are pipes, tins or buckets sunk into the ground, and are most effective when used in tandem with a drift fence to direct animals into the buckets (Corn 1994). The pitfall trapping design varies with target species, ranging from a single, long fence (e.g., 50 m) to several shorter fences (e.g., plus-shaped design). Pitfalls that are greater than 30 cm in depth which are set on both sides of the drift fence and which run along a transect of greater than 15 m are the most productive (Osborne 1985; Braithwaite 1983; Friend 1984).

Advantages of this method include: (1) the detection of secretive, cryptic, and fossorial species which are otherwise very difficult to detect; (2) elimination of observer bias, which confounds comparisons of capture rates using many other techniques (but see other bias in Melbourne, 1999); (3) pitfalls capture animals for up to 24 hours a day; and (4) pitfalls can capture animals without the survey team being on site, freeing the team to engage in other capture methods. Disadvantages include: (1) this style of trapping is not suitable for animals which are arboreal; and (2) installing the fences is labour intensive and somewhat expensive.

Pitfall Survey Design and Sampling Interval for the Present Study

The pitfall line design in the present study was two fences installed on each of the ten farms. However, one line was destroyed by stock (Hall Farm) and another line was not installed due to logistical problems (King Farm). To investigate the influence of different farm habitats on biodiversity, pitfall lines were deployed into or along a range of habitats, including rice bays, dry crops (e.g., canola, fallow), vegetation remnants, along canals, and near farm dams. In some cases a fence was installed between two or more of these habitats. Table 1 lists the habitats of each fence at each farm.

Care was taken to run pitfall lines of all farms simultaneously, due to the influence of weather conditions on detection and capture rates of animals (Duellman and Trueb, 1986; Heyer et al., 1994). Thus, we used two survey teams (1-3 persons), each of which checked fences at roughly half of the farms each day. We conducted 13 standardised surveys, each 1-3 weeks in duration, and lines were checked daily for animals during this time. Occasionally, adverse weather conditions meant that lines were checked after two days.

Fence Design and Construction

In the present study we used a single 50 m fence design for standard sampling on farms. Fences were constructed of silt fencing material and wooden stakes (these can be purchased with stakes already attached or with stakes separate). Fence bottoms were buried 12 cm deep to prevent animals from easily tunneling under the fence.

Each fence contained six pairs of buckets at 10 m intervals to capture small animals (one bucket on each side of the fence; Fig. 2). We used 20 L plastic buckets, and we placed a sponge at the bottom of each trap to provide captured animals a refuge against overheating and predators.

A funnel trap was placed at each end of each fence to capture larger animals that could climb out of the buckets (e.g., snakes, larger lizards; Fig. 2). Funnel traps were constructed of wire mesh (with 1 cm holes), and consisted of a tube (1 m long × 0.3 m diameter) and a funnel, also constructed of wire mesh. Traps were covered with pieces of linoleum to prevent animals from desiccating.

An additional trapping gird was established at the pseudocontrol site (Old Coree) in the spring/summer 2002/2003. The grid employed six parallel lines of six traps such that all traps were spaced 20 m apart. Traps in the grid consisted of 20 L buckets with a 3 m section of aluminium fencing separating each bucket. This design specifically targeted the narrow-nosed planigale (Planigale tenuirostris).

Figure 2. Photograph of a red-bellied black snake (*Pseudechis porphryiacus*) in a funnel trap employed at the end of a pitfall fence designed to capture small reptiles, amphibians, and mammals.

Field Techniques – II. Visual Encounter Survey (VES)

VES as a Detection Technique

A Visual Encounter Survey (VES sensu Crump and Scott, 1994) involves field personnel walking through an area for a prescribed time period, systematically searching for animals. This technique is also known as the 'time-constrained search' (Campbell and Christman, 1982; Corn and Bury, 1990). An example of a VES is walking through woodlands searching for reptiles and frogs under logs, bark, or other cover objects.

Advantages of this method are: (1) the technique is flexible and can be modified to detect a number of different species; (2) trained surveyors with experience in finding particular species or groups of species can often detect them quickly and efficiently using this method; and (3) the method is inexpensive, requiring little or no materials. Disadvantages include: (1) observer bias is common; an experienced herpetologist for example will find considerably more species and individuals than an inexperienced surveyor (Crump and Scott, 1994); and (2) the technique is not generally effective for species that reside underground or in the canopy (Crump and Scott, 1994).

VES in the Present Study

VES's during the present study included searching for birds, reptiles, amphibians, and mammals, both during the day and night, by searching the canopy and ground with binoculars, searching under cover objects (e.g., bark, logs, and hollow trees), searching aquatic habitats, inspecting (old) buildings, and general walk searches along farm roads and other habitats.

We did not attempt to standardise VES's to compare capture rates among farms because of logistical concerns, observer bias, and variation in habitat types and quality among farms. Thus, we attempted to uncover as many species as possible to satisfy the objective of an inventory, rather than compare absolute numbers among farms. Although this decision may result in some sacrifice in understanding what on-farm features drive biodiversity, standardising the technique would compromise the completeness of the inventory. In summary, to achieve the sometimes competing objectives of maximising an inventory and revealing factors underpinning it, we chose to standardise trapping, but use the VES opportunistically. However, where possible we employed a similar search effort (VES) across farms.

Field Techniques – III. Bird Counts

Bird Counts as a Detection Method

The most common technique for surveying bird diversity is bird counts. Surveyors count birds based on sightings and/or calls. Binoculars are commonly used to facilitate identification. Surveys are generally standardized by restricting surveys to a certain path or length of search time.

Bird Counts in the Present Study

To assess the relationship between flooded (irrigated) rice bays and bird diversity, we conducted standardised monthly bird counts on each farm. A modified version of the area search method was used (Loyn, 1986). We searched all or a subset of rice bays of each farm from a vehicle to obtain a total count of birds utilising rice bays (or dry paddocks prior to flooding). Binoculars facilitated the surveys, which were visual only. We standardized survey path and duration for each farm. Surveys were conducted between 0700 and 1000 to control for the influence of time of day on bird activity.

Field Techniques – IV. Frog Call Surveys

Call Surveys as a Detection Method

In most frog species breeding males use species-specific calls to advertise their presence to females and to other males. Call surveys exploit this habit by sampling calling frogs during the breeding season. This technique is useful because calling frogs are nearly always heard when within the observer's hearing range, while detecting frogs within the observer's visual range can be difficult due to small size, colouration, weather, position, microhabitat, and the observer's search image (Zimmerman, 1994). Species identification is facilitated by the use of frog call cassettes (Appendix 1).

Another advantage is that call surveys are very efficient for recording the presence of calling frogs because both arboreal and ground dwelling species can be surveyed simultaneously (Zimmerman, 1994). Disadvantages are that (1) the technique is useful only during the breeding period of each species; and (2) the observer must learn to identify the species-specific frog calls.

Call Surveys in the Present Study

In the present study we conducted frog call surveys opportunistically during each sampling trip. In particular, we conducted more extensive call surveys after rainfall, when calling activity is highest (Crump and Scott, 1994). We conducted call surveys on foot and from a vehicle, and these surveys were often combined with VES's. The latter surveys involved slowly driving along farm roads at night with the windows down, listening for frog calls.

Field Techniques – V. Small Mammal (Elliot) Trapping

Elliot Trapping as a Detection Method

Elliot traps are an effective means of trapping small mammals, and have been used extensively in mammal surveys in Australia (e.g., Stewart, 1979; Emison et al., 1983).

Elliot Trapping in the Present Study

In the present study we trapped for 10 nights on each farm, and employed 20-60 elliot traps per night, baited with a mixture of oats, honey, and peanut butter. Traps were set in a variety of habitats and placed in shade to prevent overheating. Traps were invariably set in the evening and checked in early morning.

Field Techniques – VI. Specialised Turtle Surveys (Hoop Trapping and Muddling)

Turtle Surveys as a Detection Method

Most turtle species can be readily detected using baited hoop traps (Fig. 3; Legler, 1960). Hoop traps are cylindrical traps made of metal or wooden hoops (e.g., 1 m dia) and netting, and include funnels at each end. Traps are baited (usually with sardines or cat food, but any meat will usually suffice), employed into aquatic habitats, and checked regularly for captures. Floating traps can be checked once daily, but submerged traps should be checked every 20 minutes to prevent drowning. Additional turtle surveys can be conducted by muddling (or noodling). This method involves walking in the water and feeling for the hard shells of turtles with feet and/or hands (Fig. 4).

Turtle Surveys in the Present Study

In the present study we used both hoop trapping and muddling to detect turtles. In the former we employed traps baited with sardines into irrigation channels, farm dams, and rice bays (Fig. 3). The latter method was particularly effective in the rice bay tofos (small channels along the perimeter of rice bays). In the irrigation channels we used a seine to concentrate turtles into a smaller area prior to muddling (Fig. 4). However, the use of a seine alone was not possible in most areas of the channels due to thick aquatic vegetation.

Figure 3. Hoop trap used to capture turtles in a supply channel on the study farm. Traps were baited with a can of sardines to attract long-necked turtles (*Chelodina longicollis*).

Figure 4. ‘Muddling’ for long-necked turtles (*Chelodina longicollis*) in a supply channel. Turtles were first concentrated using a seine, and then captured by feeling for their shells with hands and feet.

Field Techniques – VII. Specialised Bat Surveys (Harp Trapping)

Harp Trapping as a Detection Method

Harp trapping is a favoured technique for unambiguously detecting the presence of microbats, and trap design involves modifications of early designs (Tuttle, 1974; Tidemann and Woodside, 1978).

Harp Trapping in the Present Study

In the present study we used harp trapping for a brief period in 2002. Our original survey plan did not include bat trapping, but the severe drought in 2002 prompted us to survey for animals less affected (and thus, easier to detect). We trapped for ten nights. The trap was set just before dark and checked early the next morning. We used one homemade harp trap, which was placed in and around an old building at the pseudocontrol site (Fig. 5).

Figure 5. Harp trap used to capture bats at an abandoned house on the ‘pseudocontrol’ farm, Old Coree.

Other techniques – VIII. Questionnaires and ‘Talking to Farmers’

It was recognised that farmers have an intimate relationship with their land including knowledge of many species of conspicuous animals residing or moving through their land. This knowledge in many cases eclipsed that of our field team, given that farmers spend considerably more time on their farms than the research team. Thus, we communicated with the farmers regarding animals in general, and species of particular interest. This communication included both formal questionnaires and talking to farmers about vertebrates they have seen in and around on their property over the years.

Questionnaires were designed to be highly visual and easy for farmers to answer quickly to maximise the probability of response (Appendix 2). Photographs were included of four species of interest (narrow nosed planigale, southern bell frog, inland carpet python, and sand goanna) and farmers were asked if they have seen these species on their farms. Additional questions asked farmers if they had remnant vegetation on their property, were near river corridors, and the location of the nearest town. Farmers were free to provide information anonymously, but were encouraged to give contact details in order to make a greater number of on-farm contacts in the region.

II. The Intensive Studies

A black box (*Eucalyptus largiflorens*) remnant on a rice farm.

A. The Influence of Size, Shape, and Habitat Complexity of Remnant Vegetation Patches (Black Box) on Bird Diversity

Study Area

The study area for the 'birds and remnants study' was in the Coleambally Irrigation Area (CIA). The township of Coleambally, the major population centre in the CIA, is approximately 120 m above sea level, and is located on the eastern edge of the Hay Plain, an extensive area that is largely devoid of any landscape relief. The CIA is located between the towns of Jerilderie and Griffith (Fig. 1), is roughly 79,000 ha in area, and encompasses 311 large area mixed farms and 22 small horticultural farms (Coleambally Irrigation Corporation, 2003). The irrigation area was first approved in 1957, with the first farms being balloted in 1960 (Coleambally Irrigation Corporation, 2003). The Coleambally region, along with the remainder of the Riverina Bioregion, was heavily vegetated prior to European settlement (Toscan, undated). Currently the CIA only retains around 2.5% of its woody vegetation cover (Australian Museum Business Service, 2000; Fig. 6).

This region has a long farming history, with pastoral runs initially taken up between 1835 and 1869 (Toscan, undated). These runs ranged between 20,000 to 40,000 ha and were taken up as far west as Hay (Toscan, undated). Sheep grazing emerged as the most dominant form of agriculture during the 1860s, and small scale cropping practices emerged in the early 1900s (Toscan, undated). These farming practices continued until the CIA was formed (~1960), after which irrigation farming became the dominant form of agriculture in the region.

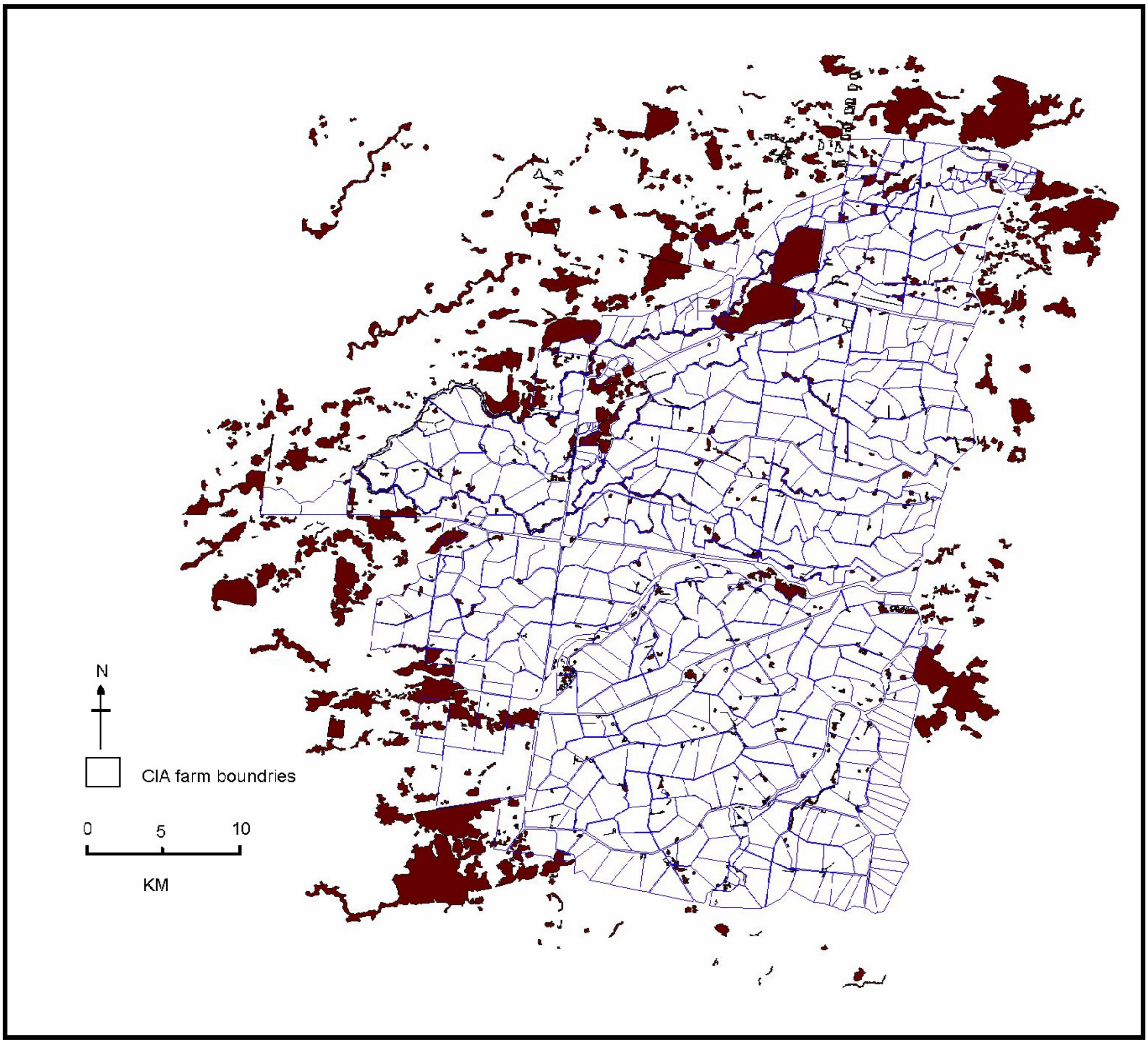


Figure 6. Woody vegetation cover within and in the immediate surrounds of Coleambally Irrigation Area (CIA). Figure created from data supplied by the Coleambally Irrigation Corporation.

Study Site Selection

Black Box Vegetation Associations

We restricted study sites to vegetative communities dominated by Black Box (Eucalyptus largiflorens) because: (1) Black Box is a major vegetation association in the Riverina; and (2) because we wanted to reduce the potentially confounding effect of vegetation communities on bird assemblages (e.g., Major et al., 2001). Black Box communities that have not been subjected to excessive grazing or clearing were typically associated with an understorey consisting of Nitre Goosefoot (Chenopodium nitrariaoeum) and Lignum (Muehlenbeckia florulenta), with lesser amounts of Thorny Saltbush (Rhagodia spinescens), Old Man Saltbush (Atriplex nummularia) and Ruby Saltbush (Enchylaena tomemiosa) (Driver and Portners, 1993; D. Bourne, pers. obs.) (Fig. 7, Fig. 8). Sites subjected to excessive grazing or clearing were largely devoid of shrubs and were dominated by native or introduced grasses and/ or forbs (D. Bourne, pers. obs.) (Fig. 9). In extreme cases sites were largely devoid of any ground cover (D. Bourne, pers. obs.) (Fig. 10). Eucalypt dieback was also evident at some sites and was widespread throughout the CIA (Fig. 9).

Figure 7. Site 206, a good quality site with a largely intact understorey. The understorey was dominated by Nitre Goosefoot (*Chenopodium nitrariaoeum*) and Old Man Saltbush (*Atriplex nummularia*).

Figure 8. Site 210, a good quality site with a largely intact understorey of Lignum (*Muehlenbeckia florulenta*).

Figure 9. Introduced grasses and forbs dominated the understorey of site 12, and the overstorey was suffering from some dieback.

Figure 10. Site 165 was almost devoid of any form of ground cover. The overstorey was relatively healthy.

Criteria for Selection of Remnant Vegetation Patches

Study sites were selected on the basis of their vegetative composition and area. These variables were chosen because they have been shown to influence the species richness and abundance of bird assemblages (Blake and Karr, 1987; Vickery et al., 1994; MacNally and Watson, 1997; Freudenberger, 1999, 2001; Watson et al., 2000; MacNally and Horrocks, 2002; Major et al., 2003). Remnant vegetation patches were chosen from a combination of aerial photos, GIS images and on-the-ground observations. Information on remnant vegetation within the CIA was obtained from Coleambally Irrigation Corporation (CIC), in the form of GIS shape files. Remnants were categorised on the basis of the dominant overstorey vegetation. At the time of the present study, there was a total of 626 remnants within, or partially within, the CIA (Fig.6). From this, the 209 remnants dominated by Black Box (E. largiflorens) were selected as potential study sites (Fig. 11). These remnants were then placed into size categories on a logarithmic scale of (1) 0.00-2.84 ha, (2) 2.85-8.09 ha, (3) 8.10-23.01 ha, (4) 23.02-65.46 ha and (5) 65.46-186.21 ha. Six sites were then randomly selected from four of these size categories, while category five restricted to four sites, yielding a total of 28 sites. However, category five consisted of only four sites. These selected sites were then graphed against a hypothetical (linear increasing) bird count, to check for a relatively even distribution from one extreme to another with respect to shape (area: perimeter ratio), isolation, and area. An unavoidable skewness towards highly isolated sites was encountered. This was overcome by transforming the data prior to analysis.

Of the 28 sites initially chosen, five sites were eliminated on the basis of (1) the denial of access permission or (2) visual inspection indicating vegetation was not optimal for the study (the aim was to select a range of structurally diverse sites). A site reconnaissance field trip conducted during 13 September 2002 – 21 September 2002 achieved this culling of sites. The remaining 23 sites were included in the study (Fig. 11). All sites were then split into four size classes for further analysis: very small <2 ha (n=4), small 2-10 ha (n=7), medium 10-50 ha (n=7) and large >50 ha (n=5). Site 210 was then added as a psuedocontrol site, as it represented an area with near natural vegetation and structure. This site was located immediately west of the CIA (Fig. 11).

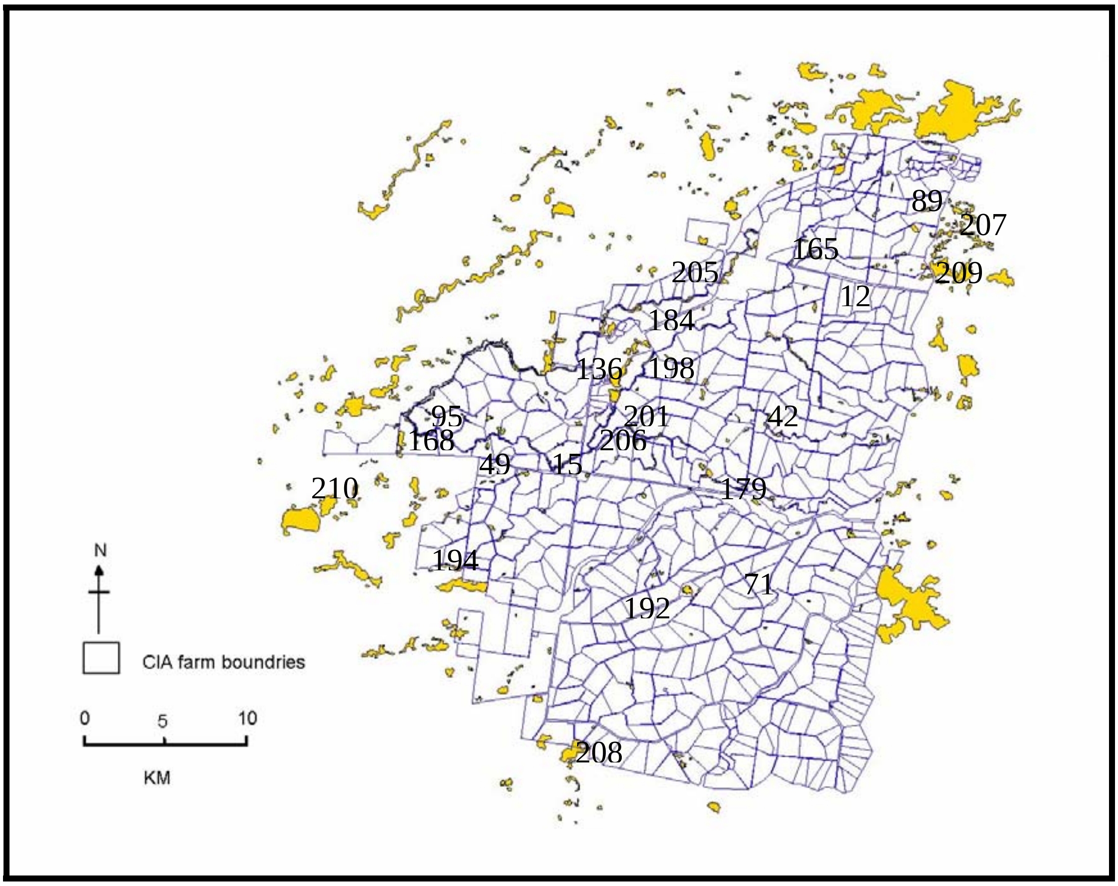


Figure 11. Cover of Black Box *Eucalyptus largiflorens* dominated vegetative communities within and in the immediate surrounds of the CIA. In contrast to the immediate surrounds of the CIA, within it there is an obvious lack of large Black Box dominated patches. Figure created from data supplied by the Coleambally Irrigation Corporation. Numbers indicate the location of farms used in the study.

Landscape and Habitat Characteristics

Landscape Characteristics

Four landscape characteristics were chosen to reflect the isolation of the patch (Table 2). These were (1) distance to nearest patch > 10 ha; (2) distance to nearest patch > 100 ha; (3) distance to nearest patch of the same vegetation > 10 ha; and (4) number of patches within 2 km. These measures of isolation were determined using a combination of GIS images, aerial photographs, and topographic maps.

Table 2. A description of the landscape and habitat characteristics measured for each remnant vegetation patch.

Abbreviation	**Description of variable**	**Unit**
Landscape variables		
DTNP >10 ha	distance to nearest patch >10 ha	m
DTNP >100 ha	distance to nearest patch >100 ha	m
DTNPSV >10 ha	distance to nearest patch of same vegetation type >10 ha	m
2 km NOP	number of patches >10 ha in a 2 km radius	number
Habitat variables		
P area	patch area	ha
P shape	patch shape	area: perimeter
CCO	canopy cover of overstorey	%
CCTS (2-4 m)	canopy cover of vegetation 2-4 m	%
CCS (0.5-2 m)	canopy cover of vegetation 0.5-2 m	%
DABH	diameter at breast height for trees >10 cm	category
Dead T	number of dead trees	number
Hollows	number of hollows	number
HG cov %	herb and grass cover	%
BG cov %	bare ground cover	%
BL cov %	branch and leaf litter cover	%
Logs	log volume	m3
Mist	mistletoe density	number
Water	water presence/absence	presence/absence
Canopy	canopy height	m
Regen	number of regenerating overstorey trees <10 cm DABH	number
Dieback	presence and extensiveness of dieback	category
Fire	fire history	category
Stems	stem density of trees >10 cm DABH	number
DABHR	range of DABH measurements across site	number

Habitat Characteristics

Twenty habitat characteristics were chosen on the basis that each could potentially influence bird diversity (Table 2). These habitat variables were recorded from a 20 x 20 m plot and a series of nested sub-plots positioned at a randomly located point and orientation within each remnant. One plot was used to asses habitat variables in patches <2 ha, two plots were used in 2-10 ha patches, and three plots were used in patches >10 ha. All variables were measured within the main 20 x 20 m plot unless stated otherwise. These characteristics were assessed between 12 November and 7 December 2002.

Patch area was gleaned from the GIS data supplied by the Coleambally Irrigation Corporation (CIC). Patch shape (as a measure of edge effects), was determined by dividing patch area (m2) by the perimeter of the patch (m). Canopy cover, cover of vegetation 2-4 m and 0.5-2 m was determined within each of the 5 x 5 m plots, yielding a total of 100 m2 (4 x 25 m2) with each 1 m2 = 1%. The cover of these vegetation layers was calculated to the nearest 5%. Diameter at breast height of trees > 10 cm was split into the following categories: category 1: 10-20 cm; category 2: 21-30 cm; category 3: 31-40 cm…….category 10: >110 cm. A tree was classified as dead if no signs of greenery were visible. Tree hollows were counted for each tree.

Relative coverage of herbs and grasses, bare ground, and branch and leaf litter were all determined within the 10 x 10 m plot. The three variables were considered non-overlapping and hence summed to 100%. Fallen timber was classified as logs if >10cm in diameter at the mid-point. Smaller diameters were included as branch and leaf litter. Log volume was determined by estimating the log length along the longest branch to the nearest 0.5 m, and then measuring the circumference at the mid-point. The number of mistletoe plants was counted within the plot to gain a density measurement. The presence or absence of water within or on the perimeter of the patch was recorded. Canopy height was recorded by measuring the height of the closest tree >80 % of the maximum tree height in the vicinity to the corner where the 10 x 10 m and 20 x 20 m plot intersect. Each tree >10 cm diameter at breast height was given a dieback index modified from Gibbons (In prep.): 1 = <30 % dieback, 2 = 31-60 % dieback, 3 = 61-99 % dieback, and 4 = dead. Fire frequency was assessed following Gibbons (In prep.): 0 = no evidence of fire, 1 = evidence of a non-recent fire event, and 2 = recent fire event.

Habitat Complexity Score

A habitat complexity score was developed from a subset of the habitat variables (Table 3). The scoring system employed was similar to that developed by Newsome and Catling (1979). As with their method, this system sought to simplify into a score, various habitat measurements indicative of the structural complexity and general "health" of the remnant. In contrast to Newsome and Catling's scoring system, however, negative scores and non-linear scores were applied in the present study (Table 3). Newsome and Catling's 'more is better' approach, in regards to their scoring system, was deemed inappropriate for the present study as some variables such as the degree of eucalypt dieback, would simply not benefit overall species presence or prevalence (e.g. Ford and Bell 1982).

Table 3. Variables that were included in, and their relative contributions to, the habitat complexity score that was developed for this study. See Table 2 for an explanation of abbreviations.

	Score						
Variable	*-2*	*-1*	*0*	*1*	*2*	*3*	*4*
CCO (%)	-	-	-	1-20	21-40	41-60	>60
CCTS (2-4 m) (%)	-	-	-	<10	10-20	20<	-
CCS (0.5-2 m) (%)	-	-	-	<10	10-30	31-50	>50
No. of trees over 60 cm DABH	-	-	-	1	2-3	>3-4	>4
Dead T (No.)	-	-	>4	1 or >3-4	2-3	-	-
Hollows (No.)	-	-	-	1-20	21-40	41-60	>61
HG and BL cov. (%)	-	-	-	1-30	31-60	61-90	>90
Logs (m^3)	-	-	0.001-0.099	0.1-0.299	0.3-0.599	0.6-0.899	>0.9
Regen. (No.)	-	-	-	1-10	11-30	31-50	>51
Dieback (category)	>2	>1-2	1	-	-	-	-

Bird Surveys

Seasonal Timing of Bird Surveys

Bird surveys were conducted on four separate field trips in 2002 (2–8 October, 23–28 October, 12–17 November, and 2–7 December). During each trip, each of the 23 sites were surveyed on two occasions, producing eight replications at each site. Sites were surveyed in a pre-determined order on the first replication within each trip. On the subsequent replication during the same trip, the sites were re-visited in reverse order to eliminate or reduce any biases associated with the time that the surveys were undertaken.

Design of Bird Counts

Birds were surveyed for 15 minutes in a 50 x 200 m (1 ha) plot using a modified version of the area search method (Loyn, 1986). Surveys were conducted at a randomly chosen location within the remnant. Using a random plot allocation allowed for a greater portion of the remnant to be surveyed over a number of different visits, thereby increasing the likelihood of obtaining a more complete species count for each remnant. In theory, in larger remnants this method maximised the detection of species that specialise in edge or interior habitats. Birds flying over or through the survey plot were not included in the count but were noted separately to indicate the species was present, but not necessarily utilising the habitat within the plot. Species presence, abundance and any breeding or unusual behaviour was recorded for each species found to be utilising the patch.

Diel Timing of Bird Surveys

Surveys were conducted between a half an hour after sunrise and 1000 hrs, to avoid the potentially confounding effect of change in behaviour, and thus detectability, through the course of the day. Upon arrival at a site, one minute was spent quietly standing at the beginning of the survey point prior to commencing the survey, to allow bird activity to return to normal. Surveys were not conducted on excessively windy days or days that were affected by precipitation as these conditions deflate bird counts (e.g. Major et al., 2001; MacNally and Horrocks, 2002).

Species Groups

Because habitat and landscape characteristics could have influences on some bird assemblages and not others, we divided birds into specific groups. We used ‘DECREASERS’ and ‘INCREASERS’ after Reid (1999), who designated birds as one or the other based on trends in the sheep-wheat belt of New South Wales. Birds not designated as DECREASERS or INCREASERS, we designated as ‘WOODLAND’ species or ‘OTHER’ species after Seddon et al. (2001). Taxonomic order, scientific and common names of birds follows Christidis and Boles (1994).

Bird Response Variables

We used species richness, density, and diversity as bird response variables. Richness was the total number of species observed at a site. Density was used because the survey plot was 1 ha for all sites. However, density would be interchangeable with abundance in the present study. Diversity was calculated using the Shannon-Wiener method and PRIMER (Carr 1997).

Study Design

We used a regression type sampling design to assess the relationship between landscape and habitat variables and bird response variables. Sites were placed into ‘size classes’ to facilitate statistical comparisons.

Univariate Analysis

Linear regression analysis was performed to assess relationships bird response variables and habitat and landscape variables. Linear regression analysis was also conducted to assess relationships between bird response variables and noisy miner (Manorina melanocephala) density.

Multivariate Analysis

Non-parametric multivariate techniques were used to determine observable differences in assemblages among sites and size classes based on mean bird abundance (Clarke and Warwick, 1994). Analyses used to detect these differences were cluster analysis, nMDS, SIMPER, and

ANOSIM. Cluster analysis nMDS, SIMPER, ANOSIM and PCA were all analysed using PRIMER (Carr, 1997).

Data Preparation

The data were log transformed prior to cluster analysis, nMDS, SIMPER and ANOSIM analysis. Some species were consistently more abundant than others, and the data were transformed to allow for less common species to contribute to the similarities observed between sites (Clarke and Warwick, 1994). The resulting inclusion of more species in the analysis was also deemed important as these species, while being less abundant, still interact with and influence other components of the assemblage. Less common species were also often species of conservation importance.

Clustering

A hierarchical clustering analysis using group-average linkings of Bray-Curtis similarities was conducted to assess groupings between sites and size classes based on bird mean abundance (Clarke and Warwick, 1994). A dendrogram was used to represent the level of similarity between sites. Using the hierarchical clustering analysis, similarities between sites and size classes of 50 % were superimposed onto the non-metric multidimensional scaling (nMDS) plot to define groups of sites at that level of similarity.

Non-metric Multidimensional Scaling (nMDS)

nMDS ordination technique was used to assess patterns in assemblages among sites and size classes (Clarke and Warwick, 1994). The stress level of an nMDS is a critical measure that indicates how well groupings (based on similarities) can be represented in two-dimensional space based on scatter (Kruskal and Wish, 1978). Plots with a stress of >0.15 should be interpreted with caution (Clarke, 1993).

Similarity Percentages Analysis (SIMPER)

A SIMPER analysis was conducted to assess the average similarity and average dissimilarity of bird assemblages within and among size classes (Clarke, 1993). The analysis also reveals the proportion that each species contributes to the dissimilarity among size classes and the similarity within size classes (Clarke, 1993). This proportion was presented as a percentage.

Analysis of Similarity (ANOSIM)

An ANOSIM was conducted to assess statistical differences in bird assemblages among size classes and pair-wise comparisons between size classes (defined a priori) (Clarke, 1993). A Bonferroni correction was applied to pair-wise comparisons to correct for type I errors. Therefore, an adjustment is made where the corrected level of significance: $(x') = x/k$ where: x = the predetermined level of significance and k = the number of tests performed (Rice, 1989). In the case of the present study, the adjusted level of significance was 0.8333%.

Principal Components Analysis (PCA)

A sub-set of landscape and habitat variables was selected to be included in a PCA ordination. The PCA ordination was used to assess which variables in the sub-set of selected variables formed coherent “components” that were relatively independent of one another (Tabachnick and Fidell, 1996). The PCA plot was then assessed to examine the degree of separation among sites based on the principal components that were derived and the amount of variation that the principal components explained. A Draftman’s plot was examined to assess normality, and if required, transformations were applied to variables with non-normal distribution to meet the assumption of normality.

Multiple Regression

A multiple regression employing a backward step procedure was used to assess the relationship between bird response variables (species richness and abundance) and landscape and habitat variables. This method was also used to assess the effects of landscape variables on bird species richness whilst incorporating the potentially confounding effect of patch area in the analysis. A backward step approach ensures that inter-correlated variables are not included in the final model. Normality was assessed by a visual examination of a histograms of residuals. Homogeneity of variances was assessed using a residual verses predicted plot.

Cumulative Species Curves

Projections for the expected number of species at a site were determined using a negative exponential model (Robinson, 1996). The negative exponential model was used to estimate the asymptote and approximate 95 % confidence intervals for the species accumulation curve at each site (Robinson, 1996). This analysis was appropriate as equal sampling intensity was applied at each study site regardless of patch size, allowing for non-biased comparisons (Robinson, 1996).

A long-necked turtle (*Chelodina longicollis*).

B. Habitat Utilisation, Movements, and Diet of Long-necked Turtles

Study Species

Description of the Species

The eastern long-necked turtle, Chelodina longicollis (family Chelidae), is a native species inhabiting freshwater habitats. The species is common and widespread in eastern Australia, extending from the Adelaide region through the Murray-Darling drainage system of Victoria and New South Wales, and continuing north to Charters Towers, Queensland (Cann, 1998).

Habitat Use and Movement Patterns

Within its range C. longicollis is known to inhabit a wide range of water bodies including permanent lakes, rivers, and ephemeral (temporary) habitats such as swamps, wetlands and billabongs (Chessman, 1988; Kennett and Georges, 1990). The species is also highly capable of exploiting artificial habitats such as farm dams (Parmenter, 1976; Chessman, 1978; Stott, 1987). The preferred habitat comprises water bodies that are shallow, seasonal and ephemeral, such as swamps and wetlands (Chessman, 1988). This is presumably because temporary habitats characteristically reach high densities of invertebrates, particularly after a complete cycle of drying and refilling (Swanson and Meyer, 1977; Brinson et al., 1981; Dannell and Sjoberg, 1982).

The ability of C. longicollis to reach and use ephemeral and seasonal habitats can be attributed to numerous physiological adaptations that enable it to survive hot and dry conditions during terrestrial travel. First, it has a low rate of cutaneous water loss by evaporation, which renders it relatively tolerant to desiccation (Chessman, 1984a). Second, it has the ability to absorb free-standing water through its cloaca (Chessman, 1984a). Third, it can maintain a large head-body temperature gradient, in which the head is kept much cooler (Webb and Johnson, 1972). The maintenance of this gradient is important because in turtles, head temperature is more directly involved in eliciting lethal heat spasms than body temperature (Webb and Witten, 1973). Fourth, it has the ability to aestivate (dormancy that enables survival during periods of extreme heat and/or drought) and can withstand at least three consecutive months without food or an aquatic medium (Chessman, 1983). Finally, this species secretes a foul-smelling substance from glands on the plastron in response to being handled or disturbed, presumably to deter would-be terrestrial predators (Chessman, 1978; Cann, 1998), although no direct evidence has been presented to support this.

Many species of freshwater turtles display movement patterns whereby individuals make annual movements to exploit seasonally available habitat s. They then retreat back to the more permanent water bodies once these habitats begin to dry (Sexton, 1959; Moll and Legler, 1971; McAuliffe, 1978; Parker, 1984; Chase et al., 1989; Graham, 1995; Reese, 1996; Bodie and Semlitsch, 2000). This behaviour is typified by C. longicollis. Previous studies have shown that this species has great aptitude for terrestrial overland movements between habitats (Parmenter, 1976; Chessman, 1978; Kennett, 1987; Stott, 1987; Kennett and Georges, 1990). In particular, C. longicollis has a marked propensity for making terrestrial movements to exploit both seasonal and ephemeral (temporary) habitats such as billabongs and swamps (Kennett and Georges, 1990; Chessman, 1978). This is presumably due to the characteristically high productivity of ephemeral and seasonal water bodies (Swanson and Meyer, 1977; Brinson et al., 1981; Dannell and Sjoberg, 1982). For example, in the Murray Valley of NSW, Chessman (1978) found that C. longicollis made seasonal movements to colonise lagoons that were re-filled annually by spring rains. Once the water level fell below 60 cm, turtles abandoned this habitat and retreated to more permanent water bodies (Chessman, 1978). Kennett and Georges (1990) found a similar pattern at Jervis Bay but over a larger temporal scale of many years. Their study found that C. longicollis made terrestrial journeys of approximately 2.5 km to reach large and highly productive swamps once they filled every several years. Turtles remained in the swamps as long as there was sufficient water. In some cases, this was as long as seven years. Once the swamps dried out the turtles retreated back to the permanent lakes.

For such movement tendencies to have been favoured by natural selection, the potential benefits in terms of evolutionary fitness must outweigh the costs (Gibbons et al., 1990). The potential cost of terrestrial movement in search of ephemeral or seasonal water bodies includes increased chance of predation, desiccation and misadventure as well as potentially high energy expenditure (Parmenter, 1976; Gibbons et al., 1990). Deaths caused by such factors have been noted previously in C. longicollis (Parmenter, 1976; Kennett, 1987). The benefits include improved growth, survival, and reproductive success (Gibbons et al., 1990; Sutherland, 1996). This is primarily due to the characteristically high productivity of temporary water bodies, particularly after drying and then refilling (Swanson and Meyer, 1977; Brinson et al., 1981; Dannell and Sjoberg, 1982). Furthermore, these water bodies usually lack fish and other species of turtle that may act as competitors and/or predators (Chessman, 1978; Bodie and Semlitsch, 2000). The study by Kennett and Georges (1990) showed that by moving to the ephemeral swamps C. longicollis obtained significantly increased reproductive output and growth. A key finding of their study was that this pattern of movement was essential for the survival of populations because low productivity combined with high densities of turtles in the lakes did not enable growth or reproduction. Thus, movements by C. longicollis to exploit temporary water bodies can yield significantly increased fitness benefits for individuals, and in some cases may be essential for their survival (Kennett and Georges 1990).

Movements and habitat use of turtles can be influenced by a variety of factors, including: the appearance of newly available habitat (Sexton, 1959; McAuliffe, 1978; Kennett and Georges, 1990), food availability (Chessman, 1978; Pluto and Bellis, 1988; Shubauer et al., 1990; Brown and Brooks, 1993), temperature and season (Lovich, 1988; Rowe and Moll, 1991; Monschein and Lewis, 1993; Brown et al., 1994; Harrel and Allen, 1996; Nieuwolt, 1996), and sexual strategy (Morreale et al., 1984; Parker, 1984; Shubauer et al., 1990; Lovich et al., 1992; Brown and Brooks, 1993). These factors are addressed in the present study.

The Effect of Human Development on the Habitat of C. longicollis

Agricultural and urban development has had adverse effects on the natural habitat of C. longicollis. Primary production industries and urban populations require a continual supply of water, resulting in major water extractions from our natural riverine ecosystems (Lemly et al., 2000). This has resulted in a reduction in the total water discharge and flood frequency in many riverine ecosystems (Lemly et al., 2000; Quinn et al., 2000). As a result, the favoured natural habitats of C. longicollis, including billabongs, backwaters, and wetlands, have been reduced in both size and frequency of occurrence (Lemly et al., 2000; Reid and Brooks, 2000). Many swamps that once contained turtles are now too transitory to sustain them (Beck, 1991).

This decline in availability of natural habitat may have been offset by the construction of numerous dams and impoundments for agricultural use. The provision of adequate supplies of water for livestock was essential for pastoral expansion into the drier parts of Australia (Landsberg et al., 1997). Today there are over four million farm dams in Australia (Timms, 1980). Furthermore, they are in relatively close proximity. Few areas of potentially productive rangeland are farther than 10 km from an artificial water source, and most are much closer (Landsberg et al., 1997). The ecosystems provided by dams are relatively simple but can be highly productive (Timms, 1980). Chelodina longicollis is highly capable of using farm dams , and as a result, can be found in the majority of farm dams within its range (Parmenter, 1976). The provision of new habitats such as dams in landscapes that may have otherwise been unsuitable, has potentially increased dispersal, abundance, and distribution in the species (Beck, 1991).

Within agricultural landscapes, long-necked turtles have also been observed to use irrigation channels (Beck, 1991). Channels may provide corridors that can facilitate movement into areas which may have been previously inaccessible due to physical barriers such as extensive stretches of dry land. Beck (1991) attributed the presence of C. longicollis in the Wimmera River, and Hindmarsh and Albacuta Lakes to the connective corridors provided by irrigation channels.

Diet

The diet of C. longicollis is exclusively carnivorous, wide-ranging and opportunistic (Parmenter, 1976; Georges, 1982; Chessman, 1984b). Turtles will generally consume any animal of edible size encountered, including aquatic and terrestrial macroinvertebrates, nekton, carrion, and tadpoles (Parmenter, 1976; Chessman, 1984b; Georges et al., 1986; Cann, 1998). When foraging, C. longicollis employs both sit-and-wait ambush and active searching strategies (Parmenter, 1976). It feeds by a ‘strike gape and suck’ method (described in Parmenter, 1976) allowing acquisition of fast moving prey (Georges et al., 1986).

Study Site

This study was undertaken on a 125 ha rice farm 2.5 km south of Finley, and is within the Murray Valley Irrigation Area (Fig. 1). The properties adjacent to this farm comprised four rice-growing farms and one dairy farm. All of the rice-growing farms grow wheat in rotation with, or around rice. This study site was selected because C. longicollis was believed to be common, and because the property layout and farming practices were considered representative of rice farms in the district. The farmland in and around the study site was characterised by improved pastures. Remnants of natural vegetation were rare near the site, and natural water bodies were absent. The major aquatic habitats in the district were irrigation channels, farm dams and seasonally available rice bays (Fig. 12).

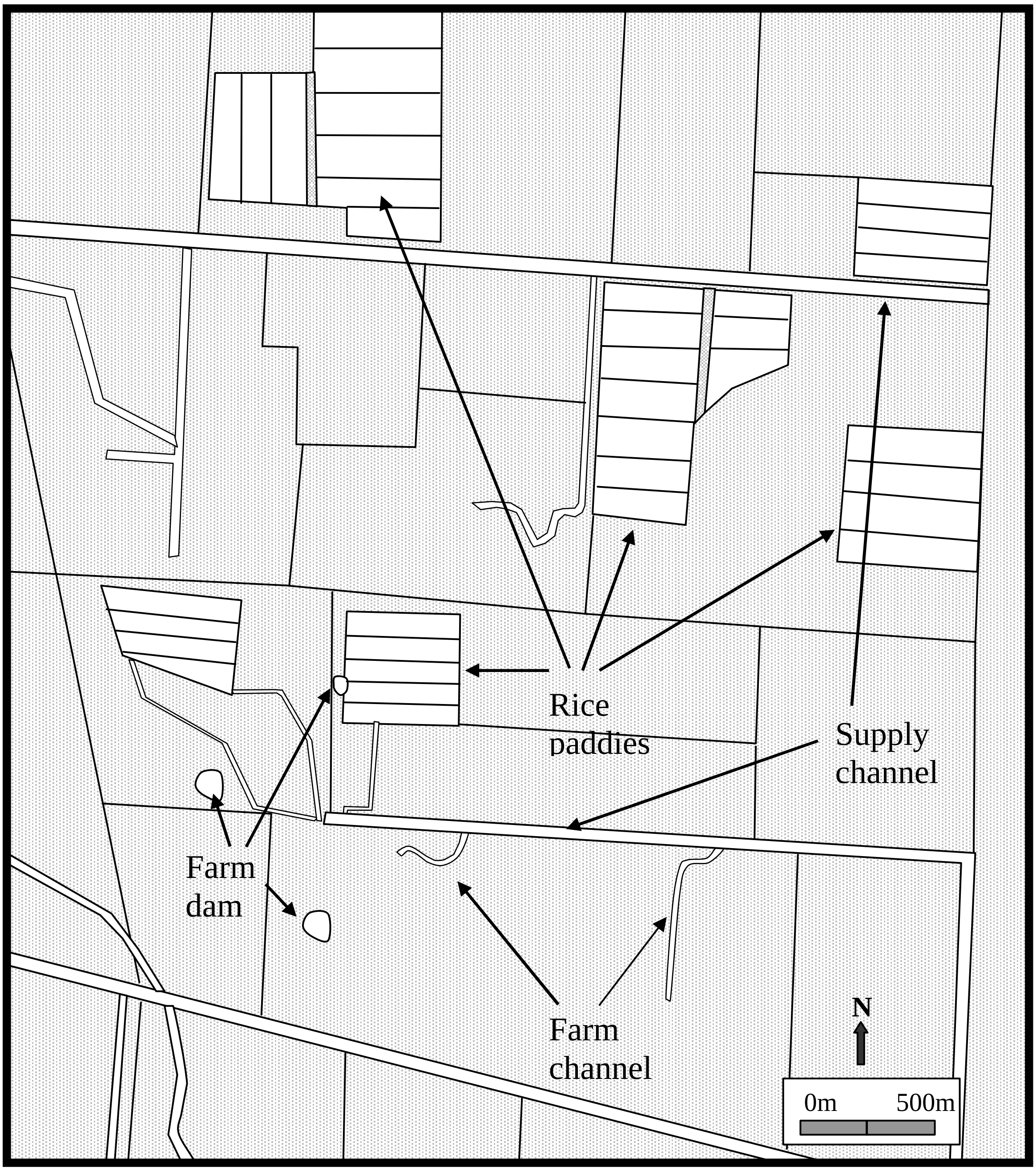


Figure 12. Diagram of the study site. Unshaded areas represent aquatic habitat, including rice paddies, supply channels, farm channels and farm dams. Grey shading represents terrestrial habitats including fallow, wheat crops and pasture. Thin black lines mark major fence lines.

Study period

A total of seven monthly field trips of 10-14 days duration were conducted on the farms between 5 September 2001 and 28 March 2002. At least two weeks separated each sampling period.

The Typical Riverina Rice Farm

The following information on rice-growing is based on information provided by the Rice-Growers Association of Australia. The average size of an Australian rice farm is around 400 hectares. In the Murray region, no more than one third of each farm is sown with rice each year, and there is a total hydraulic load limit of four mega litres per hectare. A paddock will typically contain rice for two consecutive years followed by dry field crops or pasture for the next two or three years. Cultivation of rice crops usually begins in early September. Banks of soil are formed to create levees around each rice bay. Fresh water is released from irrigation supply channels managed by Murray Irrigation Limited to flood the rice bays. The depth of the water in the rice fields is then usually maintained at 15-25 cm (depending on growing conditions) for the remainder of the growing season. Approximately 120 kg of seed is sown per hectare. This results in around 300 plants per square metre. The rice seeds are either dropped into the bays by aircraft (aerial seeding) or planted directly into dry soil by tractor drawn machinery (sod seeding). After the bays are flooded, a chemical treatment is applied to prevent pest damage to crops. No chemicals are applied for the last 100 days of the rice cultivation. Once the rice is ready to harvest the rice bays are drained and the remaining water is allowed to evaporate. Harvesting usually occurs from March to May and is fully mechanised using 'headers'.

Description of Habitats

Rice Bay Habitats

Rice bays are characterised by two different habitat types. These are the ‘rice field’ and the ‘rice tofo’ (Cohen et al., 1994) (Fig. 13). Rice tofos are small drainage depressions that mark the outer boundary of each rice bay. They are about 2 metres wide and approximately 40-50 cm deep. Tofos are created when soil is excavated to form levee banks around the rice crop. The ‘rice field’ constitutes the levelled bay upon which the rice crop is grown, with water depth shallower at 10-15 cm in depth. These two habitat types share the same water, are directly adjacent and are therefore not independent. However, they differ markedly in depth as tofos are 20-30 cm deeper than the rice field. Furthermore, rice does not generally grow within the tofos (Fig. 13). Hereafter, the term ‘rice bay’ will be used to describe the whole rice habitat including both the rice tofos and the rice field.

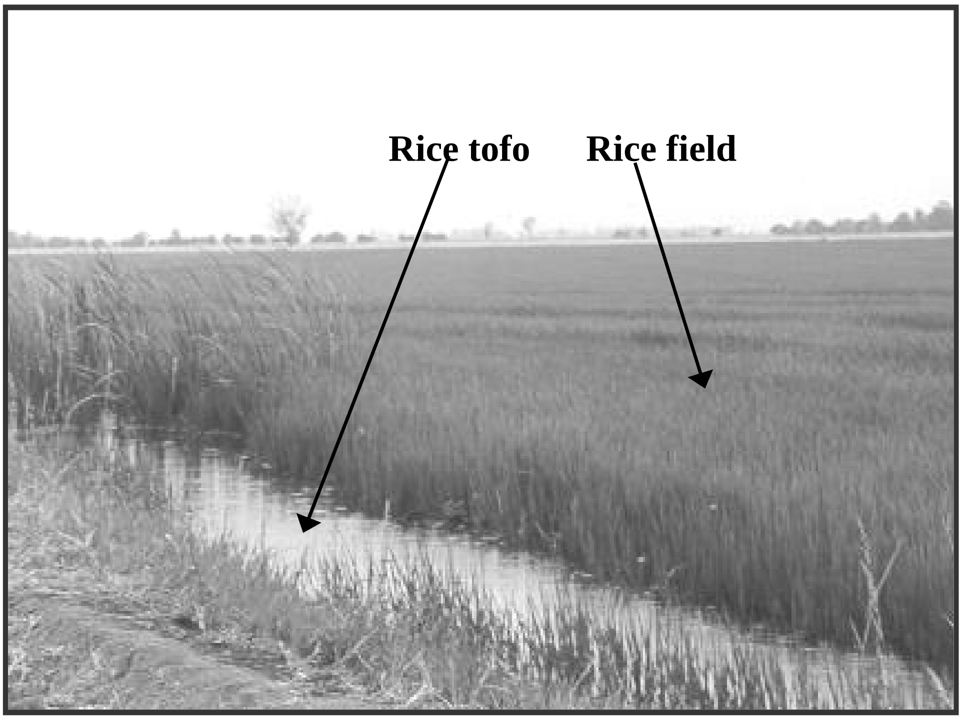


Figure 13. Photograph of the edge of a rice bay. The rice field is where the rice crop is grown and the rice tofos are depressions that extend around the inner rim of each rice bay.

Rice Bays at the Study Site

At the time of this study, five rice bays were cultivated on the Marsden farm (study site). The bays were adjacent to one another and each was approximately 440 m long and 75 m wide. These bays were flooded from 29 September to 4 October 2001. The rice seeds were aerially sewn on 5 October, 2001. From October to mid December, water depth in the rice field was maintained at approximately 20 cm. From mid-December to early January, the water depth in the rice field was reduced to approximately 5 cm to facilitate crop growth. This semi-drainage in mid/late-December is not characteristic of rice cultivation and is only used to facilitate crop growth when poor growing conditions are experienced. In January, the water was increased to approximately 25 cm and maintained at this level until mid-March. Two weeks prior to draining in late March, the water level in the rice bays was allowed to evaporate by 10 cm. The rice bay was drained over the period March 20 – 24, 2002. This drainage and harvest time in 2002 was approximately one month later than usual, the result of poor rice growth in the initial stages of cultivation (D. Marsden, pers. comm.).

Irrigation Channels: General Description and Management Practices

Throughout the irrigation district there is an extensive system of large supply irrigation channels managed by Murray Irrigation Limited (Bowmer et al., 1994) (Fig. 14; Fig. 15). Water is diverted from the Murray River into the irrigation system and delivered to farms where required (David Watts, Murray Irrigation Limited, pers. comm.). The system is operated at a constant water level throughout the season, with the flows adjusted each day depending on the demand for water by farmers (David Watts, Murray Irrigation Limited, pers. comm.). The system is operated by a gravity gate-canal system. ‘Supply channels’ as they will be referred to hereafter, exist in a broad range of sizes ranging from around 7 – 30 m in width. Depth also varies depending on the size of the supply channel but can be up to 3 m deep in the larger supply channels (David Watts, Murray Irrigation Limited, pers. comm.).

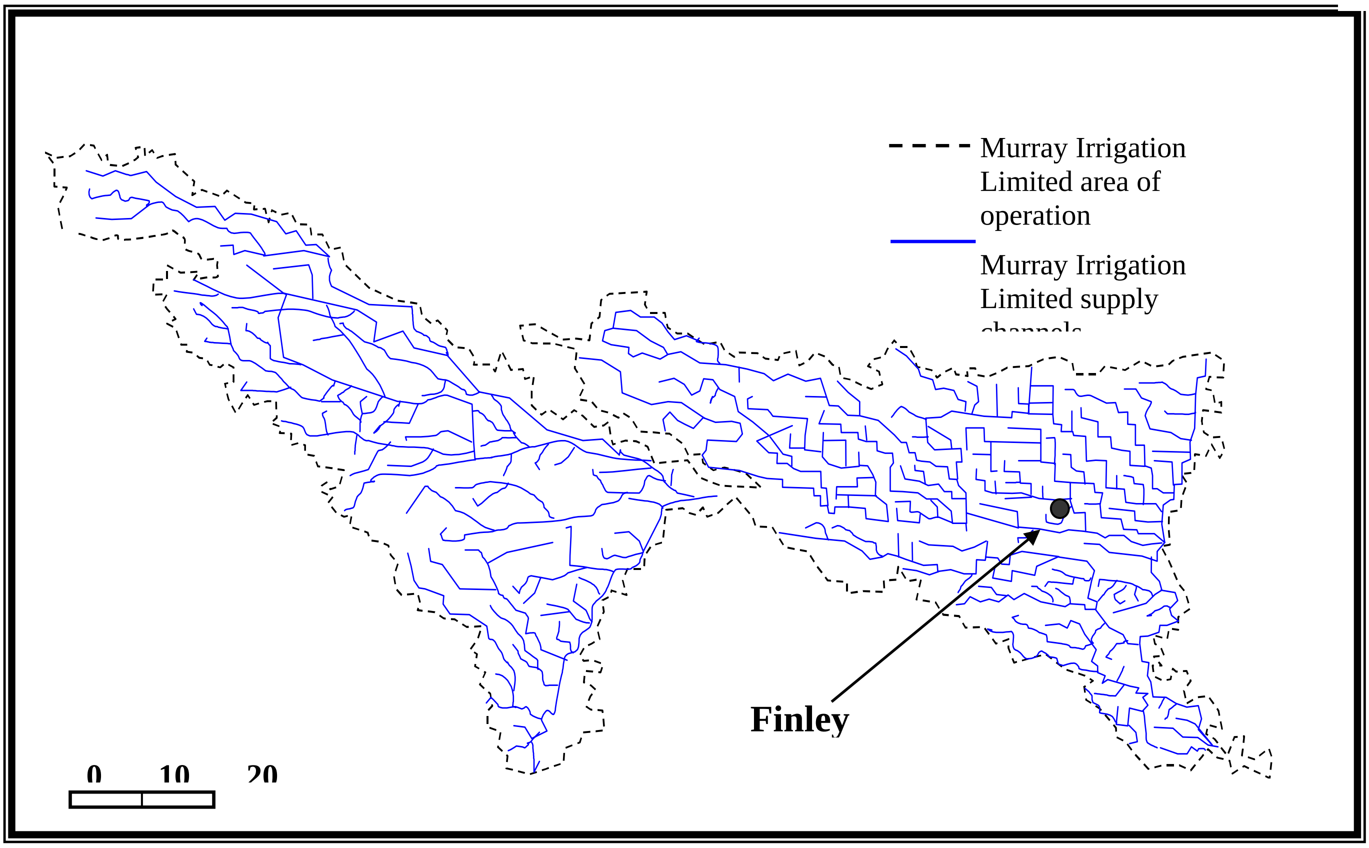


Figure 14. Network of supply channels operated for irrigation purposes by Murray Irrigation Limited in the NSW Riverina, and the relative location of Finley, which is 2.5 km north of the study site.

Figure 15. Photograph of the main supply channel in the study site. This channel supplies water to landholders and is managed by Murray Irrigation Limited.

Figure 16. Photograph of the farm channel in the study site. This channel is used to irrigate crop fields and is managed by the landholder.

Branching off the main supply channels are smaller channels operated by farmers for their own irrigation purposes. These ‘farm channels’, as they will be referred to hereafter, are much smaller than the supply channels, and are usually only a few metres wide (Fig. 16). The depth of farm channels may vary considerably depending on the irrigation needs of the farmer. Aquatic vegetation that frequently occurs in both supply channels and farm channels includes: ribbon weed (Vallisneria gigantea), arrowhead (Sagittaria montevidensis), elodea (Elodea canadensis), water couch (Paspalum distichum), curly pondweed (Potamogeton crispus), club rush (Schoenoplectus validus), slender knotweed (Persicaria decipiens), water primrose (Ludwigia pepliodes), cumbungi (Typha donimgensis), Pacific azolla (Azolla filiculoides), straight vallis (Vallisneria spiralis) and tall spike-rush (Eleocharis sphacelata).

Supply channels of the district contain a high biomass of European carp (Cyprinus carpio) (Bowmer et al., 1994). Other fish species that have been observed in the supply channels of the district include, redfin (Perca fluviatilis), oriental weather loach (Misgurnus anguillicaudatus), macquarie perch (Macquaria australasica) and silver carp (Hypophthalmichthys molitrix) (Bob Patton, Murray Irrigation Limited pers. comm.). A species of freshwater crustacean *(Cherax destructor*) is also common within the channels.

Aquatic vegetation in supply channels often reaches high densities (Bowmer et al., 1994; Bowmer et al., 1998). This can impede water flow and reduce the capacity of supply channels and reservoirs (Beyers and Carlson, 1993; Bowmer et al., 1994). To counter this, irrigation companies employ various methods of weed control (David Watts, Murray Irrigation Limited, pers. comm.). The two main methods used to control aquatic weeds are desiccation and poisoning. The supply channel network is drained annually during late May to early June and then refilled in early August. Desiccation, winter frosts and applied herbicides kill most weeds during this period. The extent to which the system dries out varies but is dependant on the rainfall and evaporation rates over the period (David Watts, Murray Irrigation Limited, pers.comm.). In a very dry year approximately fifty percent of the channels become completely dry with the remainder still muddy, whereas in the wet years the channel beds rarely dry out completely (David Watts, Murray Irrigation, pers. comm.). Chemical control is used if aquatic vegetation in supply channels reaches high densities in other months of the year when the channels are in use (Bowmer et al., 1994; Bowmer et al., 1998). Acrolein is used for submerged weeds and Glyphosate (Roundup®) is used for emergent vegetation (Bowmer et al., 1994; Bowmer et al., 1998). Aquatic vegetation can also reach high densities in farm channels. Farmers frequently employ a combination of both drainage and poisoning with Glyphosate (Roundup®) to reduce weed density (David Marsden, rice grower, pers. comm.).

Irrigation Channels at the Study Site

The supply channel in the study site was approximately 7-10 m wide and 1.5-2 m deep (Fig. 15). The depth of the supply channel in the study site varied by less than 30 cm over the course of the present study (September 2001 to March 2002). This channel was dominated by thick masses of elodea (Elodea canadensis). No poisons were applied to the supply channel in the study site during the present study.

The farm channel in the study site was approximately 3 m wide (Fig. 16) with a depth ranging from 10 to 70 cm, and was dominated by ribbon weed (Vallisneria gigantea), elodea (Elodea canadensis), and tall spike-rush (Eleocharis sphacelata). In September, the water depth was approximately 70 cm. During late October, this channel was drained from approximately 60 cm to 30 cm and the aquatic vegetation poisoned with Glyphosate (Roundup®) to improve water flow. During December, water depth increased to 60 cm. From January to March depths ranged from 20 to 40 cm.

Farm Dams of the Region

Farm dams are extremely common in the region. They vary in character depending on the nature of their use and the level of stock access. They range from typical agricultural dams that are commonly devoid of surrounding vegetation with few macrophytes and high turbidity (Buckee, 1991), to dams with dense macrophytes and abundant bird life.

The Study Site Farm Dam

The farm dam in the study site had a dense cover of thick cumbungi (Typha donimgensis) in one-half, while the other half contained no emergent vegetation (Fig.17). This dam had very limited stock access and was directly adjacent to the rice bays. The water level in the farm dam varied dramatically as a result of rainfall, irrigation water input, and irrigation water out take. During January the dam dried out completely.

Figure 17. Photograph of the farm dam at the study site. This dam was partially choked with aquatic vegetation called cumbungi (*Typha donimgensis*). The photo illustrates the close proximity of the dam to the rice paddies.

Capturing, Measuring and Marking Turtles

Turtles were captured in two ways: by trapping and by muddling. Trapping was conducted using hoop traps (after Legler, 1960) and commercially available Yabby traps (CTL traps made by Tacspo™). Traps were each baited with a can of sardines (after Parmenter, 1976). During each field trip, all major habitat types were trapped including the supply channel, rice bays, farm channel, and farm dam. Between seven and 10 hoop traps were set in the deeper supply channel, approximately

30 m apart. Yabby traps were used in the farm dam, rice tofos and farm channels because they are better suited for shallow water than hoop-traps. During each field trip, 2-3 yabby traps were set in the farm dam, 10-15 in the rice tofos, and 4-8 in the farm channel. Traps were checked 1-2 times daily.

Muddling was conducted in the rice tofos from November to March. This technique involves wading through the water using the hands and feet to detect turtles, and then capturing them by hand (Parmenter, 1976). The complete length of all tofos surrounding each of the five rice bays on the Marsden farm was muddled once each month.

Each captured turtle was marked by shell notching using a binomial system (Cagle, 1939). This entailed cutting notches in the centre of selected marginal scutes of the carapace to create unique combinations of numbers. Notches were made with a small, serrated knife.

The weight of each turtle was recorded to the nearest 0.1 ± 0. 05 g using an electronic scale. The straight-line maximum of the plastron length (PL) and carapace length (CL) were measured to the nearest 0.1 ± 0. 5 mm with vernier callipers.

Sex Classification

In most turtle species there are various external morphological features that can be used to distinguish between the sexes (Alderton, 1988). Although morphological differences exist for C. longicollis, they are subtle, making it difficult to determine the gender of this species. Parmenter (1976) identified two criteria for differentiating between the sexes of C. longicollis. These were: (1) ‘males often have a concavity in the pelvic region, when the turtle is viewed upright from the rear, whilst females often have a convexity’; and (2) ‘the vertical displacement of the plastron and carapace at the anal notch is proportionately larger in females than males.’ Until the present study, there has been no quantitative measure of either of these parameters. We found these criteria difficult to use because overlap existed in criterion (1), and criterion (2) was ambiguous due to lack of quantitative values. Thus, an attempt was made to quantify the difference in vertical displacement at the anal notch between males and females so that turtles could be easily sexed by making a simple measurement in the field. Vertical displacement of the plastron and carapace at the anal notch was measured in all turtles larger than 10 cm (the size at which Parmenter (1976) found his criteria to be reliably accurate). This measurement was then standardised by converting it to a ratio against plastron length of the turtle (i.e., vertical displacement/plastron length). These ratios were inserted into a frequency distribution plot to determine if a bimodal distribution was present and if there were discrete ranges of VD/PL ratios for each sex. At the completion of the study the turtles used in radio-telemetry were brought back to the laboratory and examined internally with laparoscopy to (1) verify their gender, (2) verify their sexual maturity status, and (3) to test the VD/PL ratio method for sex determination.

Determining Habitat Use and Movements

Three approaches were taken to examine the habitat use and movement patterns of C. longicollis. These were radio-telemetry, mark-recapture and spool-tracking. Radio-tracking was the primary method used. Mark-recapture and spool-tracking were employed to provide extra detail and increase the accuracy of the documentation of habitat use (Bodie and Semlitsch, 2000).

Radio-tracking

Radio-telemetry is one of the most useful tools available to ecologists studying spatial dynamics of animals, as it allows individuals to be monitored intermittently over many months (Kenward, 1987; White and Garrott, 1990). This method does not require continual disturbance and handling that can influence the movements of animals (White and Garrott, 1990). It has the potential to provide unbiased data on an animal’s use of both time and space (White and Garrott, 1990; Aebischer et al., 1993). However, there are a number of key assumptions that underpin most radio-tracking studies and these need to be addressed (Kenward, 1987; White and Garrott, 1990). First, it is assumed that the individuals radio-tracked are a random and representative sample from the sex and/or age class of interest (White and Garrott, 1990). Second, it is assumed that animals with transmitters move and function in the environment in the same way as those not carrying transmitters (White and Garrott, 1990). Third, most analytical procedures using radio-tracking data assume that individual observations are independent (White and Garrott, 1990).

The present study aimed to compare habitat use between sexually mature males and females. The individuals that were used for radio-telemetry were the first seven turtles of each sex captured that were of sexually mature sizes (based on Chessman, 1978). These turtles were captured both by hand (2 turtles) and trapping (12 turtles). The two hand-captured turtles were males that were opportunistically found. One was trapped in the farm channel (No. 1) and the other walking alongside a shallow drainage channel at the north end of the Marsden farm (No. 11). The turtles obtained from traps were captured in both the farm channel (turtle No.'s 4, 5, 6 & 7) and the supply channel (turtle No.'s 2, 3, 8, 9, 10, 12, 13 & 14). Although trapping does not always yield a random sample (White and Garrott, 1990), this was the only effective method available prior to flooding the rice bays. Seining of the channels may have yielded a more representative sample but this method was not practical due to the dense aquatic vegetation within the channels.

Our sample of turtles appeared to be in a healthy condition. The sex and reproductive condition of each turtle was verified at the end of the study using of laparoscopy (after Chaloupka and Limpus, 2002). The age of the turtles was unknown as C. longicollis cannot be accurately aged by the standard method of laminar growth ring counts (Chessman, 1978).

Attachment of Radio-transmitters

It is generally impossible to prove that the attachment of radio-transmitters has no effect on the individual involved (White and Garrott, 1990). However, there are a number of precautions that can be taken to limit this disturbance (White and Garrott, 1990). The factors of primary concern are the weight of the transmitter and method of its attachment (White and Garrott, 1990). The transmitters used in this study were manufactured specifically for turtles by Sirtrack™. They consisted of a radio-transmitter and battery concealed in moulded plastic mounted on metal plates. The approximate size of this package was 3 cm x 5.5 cm. In addition, the package contained an antenna of 20 cm in length that extended from the back of the transmitter. The total weight of the package was 25g. The minimum weight of turtles used in radio-telemetry was 550 g. Thus, the total weight of the transmitter amounted to less than 5% of the body weight of the turtle. This is considered a safe maximum for radio-telemetry studies (White and Garrott, 1990; Jones, 1996; Bodie and Semlitsch, 2000). Two small holes of 5 mm in diameter were drilled into the edge of the posterior marginal scutes of the carapace with a cordless drill. Bolts and lock nuts were then used to fasten the transmitter to the shell (Fig. 18). The turtles were released at their point of capture within 48 hours. Numerous studies have used this method of transmitter attachment on freshwater turtles and reported no ill affects (Ross and Anderson, 1990; Graham, 1995; Kramer, 1995; Harrel and Allen, 1996; Jones, 1996; Magnusson et al., 1997; Trauth et al., 1998; Bodie and Semlitsch, 2000; Doody et al., 2001).

The disturbance and stress involved with the initial capture, handling, and transmitter attachment usually provokes unnatural behaviour in animals that may last for a few days after the initial attachment (White and Garrott, 1990). To eliminate any bias the first three days of radio-tracking were discarded from the analysis. There is no evidence to suggest that the attachment of transmitters to C. longicollis in this study may have influenced their behaviour in any way that would have significantly affected the integrity of the outcomes of this study.

Frequency of Relocation of Radio-tagged Turtles

Turtles were located once daily for 10 to 14 days per month from September/October though to late March. The first radio-telemetric sampling period spanned late September and early October. Hereafter, these two months will be referred to as one sampling unit called ‘September/October’. All relocations were made between 9 am and 5pm. Analysis of radio-telemetry data generally assumes independence of observations (White and Garrott, 1990). Observations can be considered independent if sufficient time has elapsed for the turtle to traverse the total length of its home range (White and Garrott, 1990). In the present study, turtles were located once daily by radio-telemetry. Although size of the home ranges was not known, it became apparent that turtles were capable of traversing up to 800 m in one day. Therefore, a maximum relocation frequency of once daily was considered sufficient in providing independent data points.

Mapping of Radio-telemetry Turtle Locations

Aerial photographs of the study site and surrounding farms in various scales were obtained from Murray Irrigation Limited and used to map turtle movements. Each time a turtle was relocated the date, time, weather, habitat type and distance from a predetermined reference point was recorded. Reference points comprised notable landmarks within the study area identified from on-ground surveys and comparisons with the aerial photographs. Such landmarks included property fences, crop field edges, roads, water gates, bridges and trees. In areas where there were insufficient landmarks to form reference points, marked stakes were placed every 100 m along the aquatic habitats. Radio-telemetry fixes were later transferred manually onto aerial photographs.

Figure 18. Photograph of a male *C. longicollis* with a radio-transmitter attached.

Accuracy of Locations

Turtles were located by following the signal of each transmitter with an H-frame Yagi hand-held aerial attached to a TR-2 Telonics receiver (Sirtrack™). Turtles were usually not visually seen due to the colour and depth of the water. Instead, their exact location was estimated by the strength of the transmitter's signal. Previous trials have shown that without the aerial attachment, the transmitter signal could only be heard if the transmitter was within a 5 m radius of the receiver. Consequently, when the signal could be heard without the aerial attachment it was estimated that the turtle was within 5 m of the receiver. An additional error arose from the estimation of turtle location with respect to reference landmarks. The locations of turtles were recorded by visually estimating the distance to the nearest landmark or stake marker. Consequently, there was also a degree of human error involved in this estimation. Based on previous trials, this error was estimated to be approximately 5 metres. The standard error for each fix was thus approximately 10 metres. This level of error was considered acceptable because the habitat types available to C. longicollis fell into discrete units with no overlap. Therefore, misclassification of the habitat being used was highly unlikely.

Mark-recapture

Mark-recapture is exceptionally useful in long-term migrational studies (Sexton, 1959; Kennett and Georges, 1990). However, this method is usually ineffective in establishing daily movement of individuals (Kenward, 1987) as turtles cannot generally be recaptured daily. In the present study, mark-recapture was employed to supplement radio-telemetry and provide additional information on habitat use and movements. It was also used to estimate the size and structure of the population. Estimation of habitat use by mark-recapture can be biased by inconsistencies in trapping. Sources of bias include changes in trap location, the number of traps set, use of different types of traps and changes in bait. Due to inconsistencies in trapping in the present study, no estimates of habitat use were made from mark-recapture data.

Spool-tracking

Spool-tracking of turtles was originally pioneered by Breder (1927) and has since been widely and successfully used on turtles including C. longicollis (Stott, 1987; Wilson, 1994; Graham et al., 1996; Carter et al., 2000). It is an effective method for investigating fine-scale movement and habitat usage not generally attainable with radio-telemetry (Stott, 1987; Carter et al., 2000). However, it is also time consuming and is only suitable in terrestrial or very shallow aquatic habitats. In the present study, spool-tracking was only used in the shallow rice bays.

The spool device used in this study was designed and successfully used on bog turtles by Wilson (1994). The device consisted of a spool of fine cotton thread coated in a rubber plastic coating (Plastic Dip™) to protect it. These spools were attached to the posterior marginal scutes of the turtle's shells via duct tape (similar to a Band-Aid attachment). The total package weighed less than 10g. Only turtles with a minimum weight of 400g were used. Thus, the total weight of the package was less than 5% of their body weight. The end of the thread line was anchored to a stake inserted at the point where the turtle was captured. The turtle was then released and as it moved around, the thread unwound, leaving a trail behind it. Approximately 24 hours later the trail of thread that was left by the turtle was followed, measured and recorded on hand drawn maps of the farm. On completion, all attachments were removed from the turtles and they were returned to their location of capture.

This method required only approximately 15 minutes of handling. Turtles used in radio-telemetry were not used for spool-tracking. Most of the turtles that were spool-tracked were stomach flushed after their initial capture (see below). Stomach-flushing may influence individual movements due to stress and consequently, the results may be biased. Therefore, the spool-tracking data were not used in the calculation of daily movement distances or in any statistical analysis. Nevertheless, spool-tracking provided useful information on habitat use within the rice bays. A total of eleven turtles were successfully spool-tracked in December, and January.

Determining Diet and Prey Preference

Stomach-flushing

To examine the diet of C. longicollis, all captured turtles that were larger than 9 cm (carapace length) were stomach-flushed (except radio-telemetered turtles; Fig. 19). Most turtles were flushed immediately upon capture, the remainder within two hours of capture. Stomach-flushing was conducted as described originally by Legler (1977) but with the use of a 12V electric water pump (after Georges et al., 1986). Legler (1977) observed turtles for 12 weeks after using this method and found no ill-effects on the subjects. Dissections completed subsequent to stomach-flushing have revealed that this method retrieves all stomach contents in a non-destructive manner with no internal damage to turtles (Parmenter, 1976; Chessman, 1978).

During the stomach-flushing process a thin, soft and flexible tube (veterinary nasal tubes) attached to the 12V submersible pump, was fed down the turtle's throat to supply a slow and steady supply of water to the stomach. Various sizes of tube (diameters 3mm, 5mm and 7mm) were used to suit the size of the turtle. The turtle was held at a 45-degree angle over a sieve lined with gauze. The water pumped into the stomach via the tube was then passed from the stomach back through the esophagus, carrying with it food items from the turtle's gut. The water with its contents fell into the sieve for collection. Pumping was continued until no further stomach contents could be eluted. The tube was reinserted and repositioned periodically when necessary. The gauze that captured the stomach contents was then removed from the sieve and preserved in 70% ethanol. The prey items in the tulle were later examined under a stereomicroscope for identification to family level where possible.

The differential rates of digestion for different food types can cause a bias in diet analysis (Georges et al., 1986). To minimise this bias, partly digested prey items in the mucous plug residing in the pyloric stomach prior to flushing, were excluded from the analysis (Georges et al., 1986).

Figure 19. Stomach flushing a long-necked turtle (*Chelodina longicollis*) for diet analysis.

Prey Abundance

Prey abundance was estimated for each major habitat type of the rice agroecosystem including the supply channel, farm channel, rice tofo, rice field and the farm dam. The farm dam was sampled in two strata. The first was the section with dense emergent aquatic vegetation, and the second the section with no emergent vegetation (Fig. 17). These two habitat types were sampled individually and termed 'dam vegetated' and 'dam open' respectively. Thus, a total of six habitat strata were sampled (supply channel, farm channel, rice field, rice tofo, dam open, and dam vegetated).

Three prey samples were collected from each habitat type in each month from November to March. Samples were collected with a vertical column sampler (cross-sectional area 0.07 m^2). Unfortunately, it was not possible to collect all the samples at the same depth due to the marked differences in habitat depth and morphology. The column sampler was placed on the substrate at a depth of 40-75 cm in all habitats except the rice field, which was sampled at its depth of 10-20 cm.

Prey organisms were collected by agitating the substrate to a depth of 20 mm by hand and then sweeping a 500-µm mesh size dip net through the water column. All material collected by the dip net was tipped into sample trays. The water in the column sample was continually sieved with the dip net until ten consecutive scoops delivered no new prey items. All macrophytes within the column were removed and included in the sample to capture any invertebrates clinging to them. The samples collected in the trays were preserved in 70% ethanol and taken back to the laboratory. Each sample was then sub-sampled to obtain an estimate of the total abundance and community composition of prey items. Sub-sampling involved spreading the entire contents of the sample over a tray divided into 100 aliquots. Individual aliquots were randomly selected from the sample. All prey items in the selected aliquots were counted and identified to family level under a stereomicroscope (× 10 magnification). Individual aliquots were repeatedly removed and sorted until a minimum of 100 organisms were obtained from each sample. The total abundance of each prey type in the sample was then estimated using both the total number of each prey type obtained in the sample and the percentage of the sample analysed to obtain that number (i.e. total abundance per sample = [number of individuals obtained in all selected aliquots / number of aliquots sampled] × 100).

Prey Diversity

Ecological diversity is a measure of community structure defined by the relationship between the number of distinct taxa and their relative abundance (Poole, 1974). Diversity of prey in each habitat type was estimated with Shannon's diversity index (Magurran, 1988), calculated at family level. This index is a measurement of biodiversity that takes into account the number of individuals as well as number of taxa and the distribution of individuals among taxa (Magurran, 1988). Indices range from zero for communities with only a single taxon to high values for communities with many taxa, each with equal numbers of individuals (Poole, 1974). Numerous other indices are available. However, Shannon's diversity index was considered the most suitable because it accounts for both abundance and evenness of the taxa present (Magurran, 1988).

The level of taxonomic resolution used in calculating the index should be chosen with respect to the ecological questions posed (Kitching et al., 1998). This study was not concerned with biodiversity per se, but rather the diversity of distinctive prey types available to C. longicollis. Consequently, family level was considered to be suitable because the relatively small morphological and behavioural differences found between genera within families probably have little impact on how the turtles respond to them as prey items. Shannon's diversity indices were calculated using the program Bio-Dap (Thomas, 1988). This program calculates Shannon's diversity indices values based formulae described by Magurran (1988).

Analyses

Statistical Procedures

All statistical analysis was performed using SAS, Statistical Analysis Systems (SAS Institute, 1996) or Statistica (StatSoft Inc., 1994-1995). All statistical analysis was carried out in accordance with procedures outlined in Sokal and Rohlf (1995). Significance was at the 95 % level or greater. Means are presented with their standard errors, unless otherwise specified.

Calculation of Habitat Use

Habitat use was estimated by calculating the proportion of radio-telemetry fixes within each habitat (White and Garrott, 1990; Aebischer et al., 1993). This was examined firstly for each individual turtle, and secondly for all turtles combined in each month. Due to the lack of a standardised method of trapping, no relative estimates of habitat usage were made from capture data.

Analysis of the Influence of Sex, Month and Reproductive Season on Movement Distances

The daily movement distance (metres/day) for each turtle was defined as the shortest aquatic distance between successive daily locations. The use of straight-line distances between successive daily locations undoubtedly underestimates the true distance moved by turtles (Webb and Shine, 1997; Carter et al., 2000). For example, a study of bog turtles (Clemmys muhlenbergii) revealed that net distances measured by spool-tracking were 6.5 times greater than estimated with radio-tracking data (Carter et al., 2000). To assess the extent to which distances may have been underestimated in this study, the actual distances travelled by turtles, as recorded by spool-tracking over a 24-hour period, were compared to the straight-line distance travelled in the same period.

The effect of sex and month on movement distances was examined with a two-way ANOVA. The effects of sex and reproductive season were also examined with a two-way ANOVA. For this purpose, reproductive seasons were defined as; mating (September/October), nesting (November to December) and post nesting (January to March), based on previous studies (Parmenter, 1976; Chessman, 1978).

Seasonal Site Fidelity Analysis

In order to determine if C. longicollis shows seasonal site fidelity or has more nomadic tendencies within rice agroecosystems, the distance of turtles from their locations of original release points was regressed against the number months since their release (see Webb and Shine, 1997; Guarino, 1998; Slip and Shine, 1998). Turtles with site fidelity should exhibit a general tendency to be relocated further away from the initial release site as time goes by (Guarino, 1998; Slip and Shine, 1998).

Diet Analysis

The composition of stomach contents samples was compared to the prey availability in the environment. The prey abundance and composition in the rice agroecosystem varied between habitats and months. Consequently, comparisons were only made between the diets of turtles caught in a particular habitat type and month, with the prey availability in the same habitat type and month. Only stomach content samples with a minimum of 10 prey items were analysed. To examine the relationship between diet and prey availability, we calculated the number of each prey type in the stomach content samples and expressed this as a percentage of the total number of prey items in the stomach samples (Georges et al., 1986). Second, for each prey type in stomach contents samples, we visually estimated its percentage of the total volume of all stomach contents in the sample (Plummer, 1981). These were then compared to the relative abundance of available prey in the habitat.

Dietary Selectivity Analysis

The program PREFER (Pankratz, 1994) was used to estimate the relative selectivity placed on individual prey types by C. longicollis. Using this program we calculated selectivity ranks for each prey type based on the calculations described in Johnson (1980). This method determines the preference of individuals for food components using availability and consumption data. It then ranks the food components (Johnson, 1980; Pankratz, 1994).

The Johnson’s method was selected because it uses ranks rather than preference indices to determine selectivity (Johnson, 1980). This accounts for situations where inclusion or exclusion of certain habitats is questionable (Johnson, 1980). Preference indices generally only indicate the relative preference for a prey type compared to the other prey types in the sample (Chesson, 1978; Johnson, 1980). One of the major problems with this is that the outcomes are critically dependent on the array of components deemed to be available by the investigator. This is often an arbitrary decision (Johnson, 1980). The program PREFER was then used to test the null hypothesis that all prey types are equally preferred. This was done with the multiple comparison procedure of Waller and Duncan (1969), which statistically tested the differences between ranks of use and availability. To further examine the relationship, a correlation of the percentage of the total abundance of all prey types in the habitat to their percentage of the total abundance in diet samples, was conducted for each of the months and habitats mentioned above.

Analysis of Prey Abundance and Diversity

The effects of both month and habitat on prey abundance were examined in a two-way ANOVA. The statistical difference in prey abundance between the rice habitats and the supply channel was examined in each month with pre-planned comparisons in a Bonferroni test. This limited number of pre-planned comparisons was selected to maximise the alpha level of significance ($p < 0.\ 05$). The difference in prey abundance between the rice and channel habitats was further examined by comparing the two rice habitats (combined) against the two channel habitats (combined) in a two-way ANOVA, followed by Tukey's multiple comparisons. The same analysis was carried out to test the effects of both month and habitat type on Shannon's diversity indices.

Detailed Results

I. The Extensive Study

A. The Inventory

Species Richness

Overall Species Richness

We found a total of 201 species of vertebrates inhabiting or using rice farms during the study (179 species detected on the 10 study farms in the 'general inventory', plus an additional 22 bird species detected in the 'birds and remnants study'). The 178 species included 149 species of birds, 23 reptile species, 19 mammal species, seven frog species, and three fish species (Table 4). Fish were not targeted in the study, but were observed opportunistically. Bats were only targeted for part of one season. Reptile species consisted of 14 species of lizards, seven species of snakes, and two species of turtles. Fifteen species of exotic or feral vertebrates were detected, including 10 mammal species, three bird species, and two species of fish.

Table 4. Species list for the extensive study, and the farms in which each species was detected. 1 = detected, 0 = not detected; Core = Old Coree, Wood = Woodside, Houg = Houghton, King = King, Step = Stephenson, Ewan = Ewan, Hoga = Hogan, Mars = Marsden, Hall = Hall, Drap = Draper. *** =** bat species were only sampled at Old Coree.

Species	Core	Wood	Houg	King	Step	Ewan	Hoga	Mars	Hall	Drap
Birds										
spotted harrier	0	0	0	0	0	0	0	1	0	0
swamp harrier	0	0	1	0	1	1	1	0	0	0
square-tailed kite	0	0	0	0	0	0	1	0	0	0
black kite	0	0	1	0	0	1	1	0	0	1
black-shouldered kite	1	1	1	1	1	1	1	1	0	1
whistling kite	1	1	0	1	1	1	1	0	0	1
little eagle	1	0	0	0	0	0	0	0	0	0
white-bellied sea eagle	1	0	0	0	0	0	0	0	0	0
wedge-tailed eagle	1	0	1	0	0	1	0	0	0	1
barn owl	0	0	0	0	1	0	0	0	0	0
boobook owl	1	0	0	0	0	0	0	0	0	0
brown songlark	0	0	0	1	0	0	0	0	0	0
azure kingfisher	0	0	0	1	0	0	0	0	0	0
wood duck	1	1	1	1	1	0	1	0	1	0
pacific black duck	1	1	1	1	1	1	1	1	1	1
chestnut teal	0	0	0	0	0	0	0	1	0	0
Australian shelduck	1	0	0	0	0	0	0	0	0	0

black swan	0	1	0	0	0	0	0	0	0	0
darter	1	0	0	1	1	0	0	0	1	0
little egret	0	0	0	0	0	0	0	0	0	1
white-necked heron	1	1	1	1	1	1	1	1	1	1
white-faced heron	1	1	1	1	1	1	1	1	1	1
Australasian bittern	0	0	0	0	0	0	0	1	0	0
nankeen night heron	1	0	1	0	1	0	0	0	0	0
great egret	1	1	1	1	1	1	1	1	1	1
intermediate egret	0	1	0	1	0	0	0	0	0	1
white-breasted woodswallow	1	1	0	0	0	0	0	0	0	0
white-browed woodswallow	0	1	0	0	0	0	0	0	0	0
black-faced woodswallow	0	0	0	0	0	0	1	1	1	0
dusky woodswallow	1	0	0	0	0	0	0	0	0	0
little woodswallow	1	0	0	0	0	0	0	0	0	0
pied butcher bird	1	0	1	0	1	0	1	1	0	0
Australian magpie	1	1	1	1	1	1	1	1	1	1
pied currawong	1	0	1	0	1	0	0	1	0	0
grey butcherbird	1	0	1	0	0	0	0	0	0	0
black-faced cuckoo-shrike	1	1	0	0	1	0	1	1	0	1
white-winged triller	0	1	0	0	0	0	0	1	0	0
masked lapwing	1	1	1	0	0	0	1	1	1	1
red-kneed dotterel	0	0	0	0	0	0	0	0	0	1
brown treecreeper	1	0	0	1	0	0	0	0	0	0
crested pigeon	1	1	1	1	1	1	1	1	0	1
peaceful dove	1	1	1	1	0	0	0	0	0	0
common bronzewing	1	0	0	0	0	0	0	0	0	0
dollarbird	0	0	0	1	0	0	0	0	0	0
apostle bird	1	0	1	0	0	0	0	1	0	0
white-winged chough	1	0	1	1	1	0	0	0	0	0
Australian raven	1	1	1	1	1	1	1	1	1	1
little raven	0	1	1	1	1	1	1	1	1	1
grey fantail	1	0	0	0	0	0	0	0	0	0
willie wagtail	1	1	1	1	1	1	1	1	1	1
magpie lark	1	1	1	1	1	1	1	1	1	1
restless flycatcher	1	0	0	0	0	0	0	0	0	0
emu	1	0	0	0	0	0	0	0	0	0
nankeen kestrel	1	0	0	1	1	1	1	1	1	1

Australian hobby	1	1	0	1	1	0	0	0	0	0
brown falcon	1	1	0	0	1	1	1	0	1	1
black falcon	0	0	0	0	0	0	0	1	0	0
brolga	0	0	0	0	1	0	0	0	0	0
laughing kookaburra	1	0	1	1	1	1	0	0	0	0
sacred kingfisher	1	1	0	1	0	1	0	0	0	0
welcome swallow	1	0	0	1	0	1	1	0	1	1
fairy martin	0	1	0	0	0	1	1	0	0	0
tree martin	0	1	0	0	0	1	0	0	0	0
whiskered tern	0	1	0	0	0	1	1	0	1	1
silver gull	0	0	0	0	0	0	0	0	0	0
superb fairy-wren	1	0	0	0	1	0	1	0	0	0
white-naped honeyeater	1	1	0	0	0	0	0	0	0	0
white-fronted chat	1	0	0	0	0	0	0	0	0	0
white-plumed honeyeater	1	1	0	1	0	1	0	0	0	0
noisy friarbird	0	0	1	0	0	0	0	0	0	0
little friarbird	1	0	0	0	0	0	0	0	0	0
blue-faced honeyeater	0	0	0	1	0	0	0	0	0	0
singing honeyeater	1	0	0	0	0	1	0	0	0	0
striped honeyeater	1	0	0	0	0	1	0	0	0	0
spiny-cheeked honeyeater	1	1	0	0	0	0	0	0	0	0
noisy miner	1	1	1	1	1	1	1	1	0	1
rainbow bee-eater	1	1	0	0	0	0	0	0	0	0
richards pipit	0	1	0	0	0	0	0	1	1	1
blackbird	0	1	1	0	1	1	0	1	0	0
jackie winter	1	0	0	0	0	0	0	0	0	0
grey shrike-thrush	1	1	0	0	0	0	0	0	0	0
rufous whistler	1	0	0	0	0	0	0	0	0	0
striated pardalote	1	1	0	0	0	0	0	0	0	0
yellow-rumped thornbill	1	0	0	0	0	1	0	0	0	0
chestnut-rumped thornbill	1	0	0	0	0	0	0	0	0	0
weebill	0	1	0	0	0	0	0	0	0	0
house sparrow	1	1	0	0	0	1	1	1	0	1
Australian pelican	0	1	0	0	1	0	0	0	0	1
red-capped robin	1	0	0	0	0	0	0	0	0	0
great cormorant	0	1	1	1	1	1	1	1	1	0
pied cormorant	1	1	0	0	0	1	0	0	1	1
little pied cormorant	0	0	0	0	1	1	1	0	1	0

little black cormorant	0	1	1	0	0	0	0	0	0	0
brown quail	1	0	0	0	0	0	0	1	1	0
stubble quail	1	0	0	0	0	0	0	0	0	0
zebra finch	1	1	0	1	1	1	1	1	1	1
tawny frogmouth	1	0	1	0	0	0	0	0	0	0
Australasian grebe	1	0	0	1	1	0	0	0	1	0
grey-crowned babbler	1	0	0	0	0	0	0	1	0	0
cockatiel	0	1	1	1	0	1	0	0	0	1
sulfur-crested cockatoo	0	0	0	1	0	0	0	0	0	0
little corella	0	0	1	0	0	0	0	0	0	0
bluebonnet	1	0	0	0	0	1	1	0	0	0
red-rumped parrot	1	1	1	1	1	1	0	1	1	1
crimson rosella	1	0	1	1	0	0	1	0	0	0
superb parrot	1	0	1	0	0	0	1	0	0	0
eastern rosella	1	0	1	1	1	0	1	1	0	0
ring-necked parrot	0	0	0	0	0	1	0	0	0	0
galah	1	1	1	1	1	1	1	1	1	1
dusky moorhen	0	0	0	0	0	0	0	0	0	0
black-tailed native hen	0	0	1	0	1	0	0	0	1	0
purple swamphen	1	0	0	0	0	0	0	0	0	0
black-winged stilt	0	1	0	0	0	0	0	1	1	1
marsh sandpiper	0	1	0	0	0	0	0	0	0	1
starling	1	1	1	1	1	1	1	1	1	1
golden-headed cisticola	0	1	1	0	0	1	0	1	1	1
clamorous reed warbler	1	0	0	0	0	0	0	0	0	0
Australia white ibis	0	1	0	1	1	1	1	1	1	1
straw-necked ibis	0	1	1	1	1	1	1	1	1	1
yellow-billed spoonbill	1	1	1	0	1	1	0	0	1	1
glossy ibis	0	0	0	0	0	0	0	0	0	0
royal spoonbill	1	1	1	0	0	1	0	0	1	0
western gerygone	1	0	0	0	0	0	0	0	0	0
silvereye	1	0	0	0	0	0	0	0	0	0
mistletoe bird	1	0	0	0	0	0	0	0	0	0
southern whiteface	1	0	0	0	0	0	0	0	0	0
bush stone curlew	1	0	0	0	0	0	0	0	0	0
lizards										
lace monitor	1	1	1	0	0	1	0	0	0	0
Gould's monitor	1	0	0	0	0	0	0	0	0	0

bearded dragon	1	0	0	0	0	1	1	0	0	0
robust ctenotus	0	0	0	0	0	0	0	1	0	0
Boulenger's skink	1	1	1	1	1	1	1	1	1	1
common dwarf skink	1	0	1	0	1	1	0	1	0	0
spotted lerista	0	1	0	0	0	0	0	0	0	0
eastern blue-tongued lizard	1	0	0	0	1	0	0	0	0	0
wall lizard	1	1	1	1	1	1	1	0	0	0
tree dtella	0	1	1	1	0	0	0	0	0	0
southern spiny-tailed gecko	1	0	0	0	0	0	0	0	0	0
tesellated gecko	1	0	0	0	0	0	0	0	0	0
wood gecko	1	0	0	0	0	0	1	0	0	0
olive legless lizard	1	0	0	0	1	1	1	0	0	0
snakes										
carpet python	1	1	0	0	0	0	0	0	0	0
eastern brownsnake	1	1	1	1	1	1	1	1	1	1
tiger snake	0	0	0	0	0	0	0	1	0	0
red-bellied black snake	1	0	1	1	0	0	0	0	0	1
curl snake	1	0	0	0	0	0	0	0	0	0
southern blind snake	1	0	0	0	0	0	0	0	0	0
prong-snouted blind snake	1	0	0	0	0	0	1	0	0	0
turtles										
eastern long-necked turtle	0	0	1	1	0	0	0	1	0	1
Maquarii River turtle	0	0	0	1	0	0	0	0	0	0
frogs										
giant banjo frog	1	0	0	0	0	0	0	0	0	0
plains froglet	1	1	1	1	1	1	1	1	1	1
southern bell frog	0	0	0	0	0	1	1	0	0	0
spotted grass frog	1	1	1	1	1	1	1	1	1	1
barking marsh frog	1	1	1	1	1	1	1	1	1	1
painted burrowing frog frog	1	0	0	0	0	0	0	0	0	0
Peron's treefrog	1	1	0	1	1	1	1	0	0	0
mammals										
echidna	1	0	0	0	0	0	0	0	0	0
narrow-nosed planigale	1	0	0	0	0	0	0	0	0	0
eastern grey kangaroo	1	1	1	1	1	1	1	0	1	0
western grey kangaroo	1	1	0	0	1	0	0	0	1	0
red kangaroo	1	0	0	0	0	0	0	0	0	0

black wallaby	1	0	0	1	0	0	0	0	0	0
goat	0	0	0	1	0	0	0	0	0	0
fox	1	1	1	1	1	1	1	1	1	1
feral cat	1	0	0	1	0	0	0	0	0	0
common brushtail possum	1	1	1	1	1	1	1	1	1	0
water rat	1	0	0	0	0	0	0	1	0	0
hare	1	0	0	0	0	0	0	0	1	0
rabbit	1	1	0	1	1	1	1	0	1	0
house mouse	1	1	1	1	1	1	1	1	1	1
narrow-nosed planigale	1	0	0	0	0	0	0	0	0	0
southern free-tailed bat*	1	0	0	0	0	0	0	0	0	0
lesser long-eared bat*	1	0	0	0	0	0	0	0	0	0
little forest bat*	1	0	0	0	0	0	0	0	0	0
chocolate wattled bat*	1	0	0	0	0	0	0	0	0	0

Species Accumulation with Effort/Time

Species accumulation curves, which illustrate the rate of addition of new species over time or survey effort, indicate that most species were found after two years (or 12 surveys), given the techniques used. Diminishing returns over time/effort were evident in all groups (Fig. 20, 21), which began to show near-saturation by the second year. The rapid rise of the mammal curve late in the study reflected the addition of bat species which were not targeted until that time (Fig. 20).

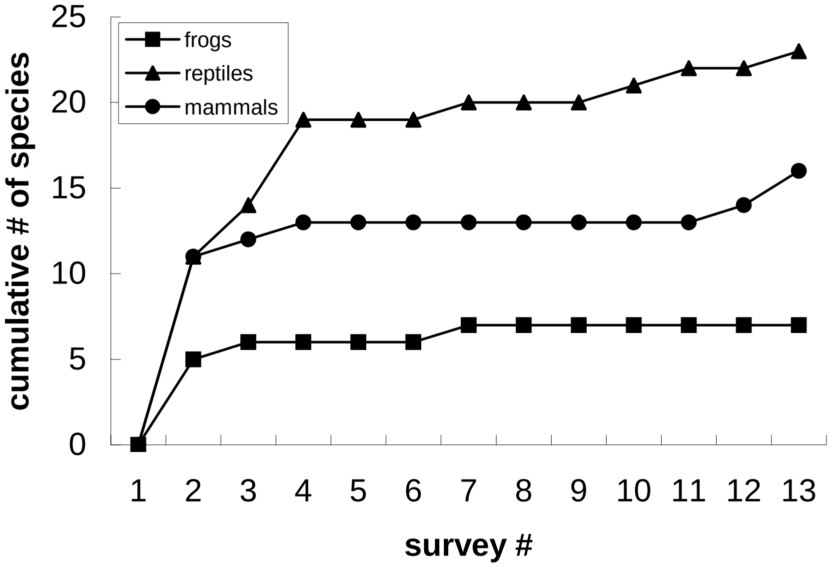


Figure 20. Species accumulation curve for frogs, reptiles, and mammals during the extensive study, showing near saturation given the techniques utilised.

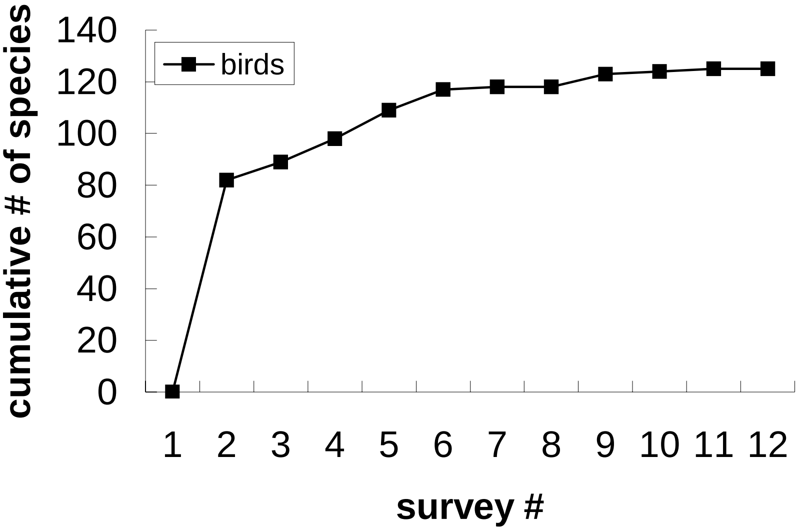


Figure 21. Species accumulation curve for birds during the extensive study, showing near saturation given the techniques utilised.

Significant Species

Threatened Species

Seven threatened species were recorded utilising rice farms during the study (Table 5), of which five were detected on the 10 study farms: southern bell frog (Litoria raniformis), brolga (Grus rubicunda), grey-crowned babbler (Pomatostomus temporalis), Australiasian bittern (Botaurus poiciloptilus), and superb parrot (Polytelis swainsonii). Two species, the painted honeyeater (Grantiella picta) and grey falcon (Falco hypoleucos), were detected on rice farms during the 'birds and remnants study' (in the CIA). We found evidence that two of the threatened species, the southern bell frog and superb parrot, were breeding on the farms. We found southern bell frog egg masses on Hogan farm, in the CIA, and a nest of the superb parrot on Woodside Farm in the MIA.

Table 5. Listed species detected during the study and their official conservation status. V = vulnerable, E = endangered, CE = critically endangered. AUS = Commonwealth of Australia (1999); NSW = Threatened Species Conservation Act (1995); VIC = Flora and Fauna Guarantee Act (1988).

Species	**Scientific name**	**Conservation status**
southern bell frog	*Litoria raniformis*	V_{AUS}, E_{NSW}
brolga	*Grus rubicundus*	$V_{NSW,}$ V_{VIC}
grey-crowned babbler	*Pomatostomus temporalis*	E_{VIC}
superb parrot	*Polytelis swainsonii*	V_{AUS}, $V_{NSW,}$ E_{VIC}
Australasian bittern	*Botaurus poiciloptilus*	V_{NSW}
painted honeyeater	*Grantiella picta*	$V_{NSW,}$ V_{VIC}
grey falcon	*Falco hypoleucos*	$V_{NSW,}$ CE_{VIC}

'Non-threatened Significant' Species

We detected several species of unlisted vertebrates considered to be rare or relatively uncommon in the area. Although not listed as threatened, these species may be in decline locally (or more generally) due to habitat loss or scarcity. Table 6 lists several species from our on-farm inventory that we feel are uncommon to rare (but not officially listed) due to rareness of habitat in the rice-growing areas.

Table 6. A sample list of species found in and around rice farms with no official conservation status, but which are uncommon, rare, isolated, secretive, or uncertain at different spatial scales. 1 = Uncommon/rare/isolated/uncertain throughout its range; 2 = Common or secure throughout its range but regionally extralimital, or uncommon/rare/isolated/uncertain; 3 = Common throughout its range and regionally common but locally extralimital, or uncommon/rare/isolated/ /uncertain; 4 = secretive with status uncertain. Designations are our hypotheses based on available information, published and unpublished.

Common name	Scientific name	Unofficial designation
narrow-nosed planigale	Planigale tenuirostris	2
black Wallaby	Wallabia bicolor	2
inland carpet python	Morelia spilota metcalfei	2
southern blind snake	Ramphotyphlops australis	4
Gould's monitor	Varanus gouldii	2-3
lace monitor	Varanus varius	3
wood gecko	Diplodactylus vittatus	3
tessellated gecko	Diplodactylus tessellatus	2-3
southern spiny-tailed gecko	Diplodactylus intermedius	2-3
spotted lerista	Lerista punctatovittata	4
olive legless lizard	Delma inornata	3
broad-shelled turtle	Chelodina expansa	1
emu	Dromaius novaehollandiae	2-3
brown treecreeper	Climacteris picumnus	2
grey-crowned babbler	Pomatostomus temporalis	3
giant banjo frog	Limnodynastes interioris	2-3

Significant Species Questionnaire

During the 2003 annual Rural Industries Research and Development Workshop, held at Old Coree, we presented attendant farmers with a questionnaire pertaining to significant species. A total of 105 farmers completed the questionnaire, although a few were only partially filled out. Twenty farmers (19%) asserted that they had seen narrow-nosed planigales on their farm in the last five years, 29 (28%) reported sighting inland carpet pythons, 31 (30%) had seen southern bell frogs, and 30 (29%) claimed to have seen Gould's monitors on their property. Seventy-five (71%) farmers reported having remnant vegetation patches on their property, and 30 (29%) reported having a wooded creek or river along, or on their property.

The questions on remnant vegetation and creek or river were designed to reinforce the validity of reported sightings of the four species, because those species were expected to be dependent on those habitats (except possibly the southern bell frog). Figure 22 illustrates the association between the presence of on-farm remnant vegetation and the number of farmers reporting the occurrence of the four significant species. Of the 29 farmers reporting sightings of the inland carpet python, 27 (93%) also confirmed the presence of either remnant vegetation patches or a creek line/river. Of these 27, all reported remnant vegetation patches and 11 reported a creek/river. Of 73 farms with vegetation remnants, 27 (37%) reported inland carpet pythons. The presence of inland carpet pythons was significantly associated with the presence of remnant vegetation patches ($\chi 2 = 6.20$, df = 1, p = 0.013), but was independent of the presence of creeks/rivers ($\chi 2 = 1.86$, df = 1, p = 0.138).

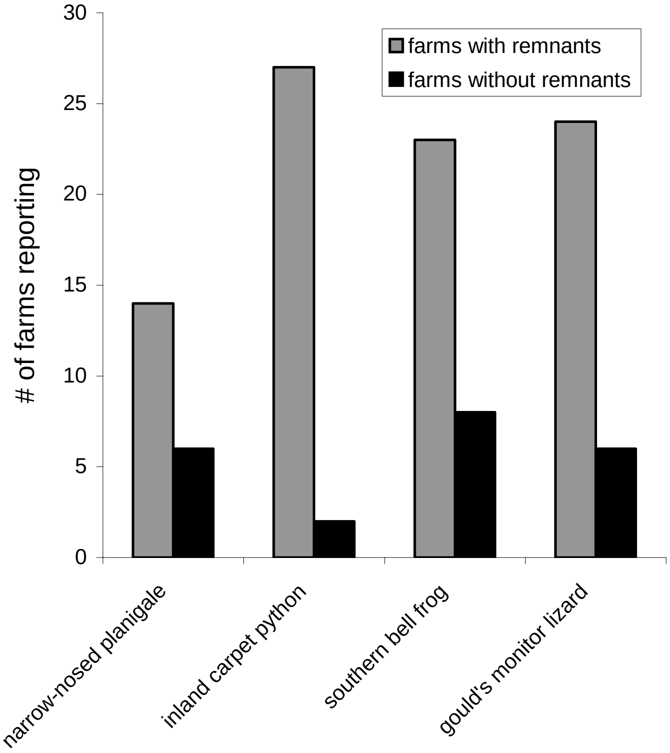


Figure 22. Association between the presence of on-farm remnant vegetation patches and the presence of four 'significant species', as reported by farmers in a questionnaire during the 2003 Rice Field Day.

Of the 20 farmers reporting sightings of the narrow-nosed planigale, 15 (75%) also confirmed the presence of either remnant vegetation patches or a creek line/river. Of these 15, all reported remnant vegetation patches and 9 reported a creek/river. Of 71 farms with vegetation remnants, 14 (20%) reported narrow-nosed planigales. The presence of narrow-nosed planigale was independent of the presence of remnant vegetation patches ($\chi 2 = 1.22$, df = 1, p = 0.269), but not independent of the presence of creeks/rivers ($\chi 2 = 4.42$, df = 1, p = 0.036).

Of the 31 farmers reporting sightings of the southern bell frog, 23 (74%) also confirmed the presence of either remnant vegetation patches or a creek line/river. Of these 23, all reported remnant vegetation patches and 8 reported a creek/river. Of 65 farms with vegetation remnants, 23 (35%) reported southern bell frogs. The presence of southern bell frogs was independent of both the presence of remnant vegetation patches ($\chi 2 = 0.051$, df = 1, p = 0.822), and the presence of creeks/rivers ($\chi 2 = 0.001$, df = 1, p = 0.971). Because the present regional distribution of the southern bell frog is believed to be restricted to isolates, and yet is likely poorly known (Australian Museum Business Service, 2000), we examined the localities of farms reporting the species. The majority of southern bell frog sightings were on farms near Coleambally (Table 7). At the time of this survey southern bell frogs were common near Coleambally (pers. obs., Australian Museum Business Service, 2000; Pyke, 2002).

Table 7. Reported occurrence of the southern bell frog (*Litoria raniformes*) on rice farms in the NSW Riverina. Data compiled from questionnaire responses from 105 rice farmers. At the time of this survey, southern bell frogs were known to be common near Coleambally (pers. obs., Australian Museum Business Service, 2000; Pyke, 2002), but unknown from the other localities listed below.

Nearest town to farm	# of farms reporting frogs	# of farms participating	% ‘occurrence’
Coleambally	12	18	67
Deniliquin	2	15	13
Finley	3	14	21
Jerilderie	4	10	40
Leeton	1	8	13
Griffith	1	7	14
Moulamein	3	6	50
Wakool	2	2	100
Blighty	1	1	100
Wagga Wagga	1	1	100
Mathoura	1	1	100

Of the 30 farmers reporting sightings of Gould’s monitor, 24 (80%) also confirmed the presence of either remnant vegetation patches or a creek line/river. Of these 24, all reported remnant vegetation patches and 11 reported a creek/river. Of 64 farms with vegetation remnants, 24 (38%) reported Gould’s monitors. The presence of Gould’s monitor was independent of the presence of remnant vegetation patches ($\chi 2 = 0.98$, df = 1, p = 0.323), and the presence of creeks/rivers ($\chi 2 = 3.14$, df = 1, p = 0.076), although the latter approached significance.

B. Factors Underpinning Vertebrate Diversity on Rice Farms

Species Richness by Farm and Comparisons Among Farms

Table 4 shows which species were detected on each of the 10 study farms. Table 8 breaks down species richness by farm. Old Coree produced the highest species richness of birds, reptiles, mammals, frogs, and total species of the 10 farms (Table 8). The total richness was lowest at Hall Farm (Table 8). The lowest species richness of birds and reptiles occurred at Hall Farm, and the lowest richness of mammals occurred at Draper Farm (Table 8). Frog species richness was equally lowest at four farms (three species at Houghton, Marsden, Hall, and Draper Farms; Table 8).

Table 8. Species richness of vertebrate groups on the 10 study farms.

Farm	Species richness				
	birds	reptiles	mammals	frogs	total
Old Coree	82	17	18	6	123
Woodside	54	7	6	4	71
Houghton	43	8	4	3	58
King	41	7	8	4	60
Stephenson	42	6	6	4	58
Ewan	45	7	5	5	62
Hogan	39	7	5	5	56
Draper	39	5	4	3	51
Marsden	34	2	7	3	46
Hall	39	3	2	3	47

The Influence of Remnant Vegetation Patches on Species Richness on Farms

The following results consider 10 farms, of which seven contained remnant vegetation (Old Coree, Woodside, Houghton, King, Stephenson, Ewan, Hogan), and three did not (Marsden, Hall, Draper). Because Old Coree was atypical and considered a ‘pseudocontrol’ (see Methods), we conducted each statistical analysis twice, once including Old Coree, and once without.

Farms with remnant vegetation patches produced more overall species than farms without remnant patches (Fig. 23, Fig. 24). This difference was only evident when excluding Old Coree, however ($F_{1,8} = 13.00$, $p = 0.008$; means = 61.0 vs. 48.7). When including Old Coree, results were insignificant ($F_{1,9} = 2.4$, $p = 0.157$; means = 69.0 vs. 48.7).

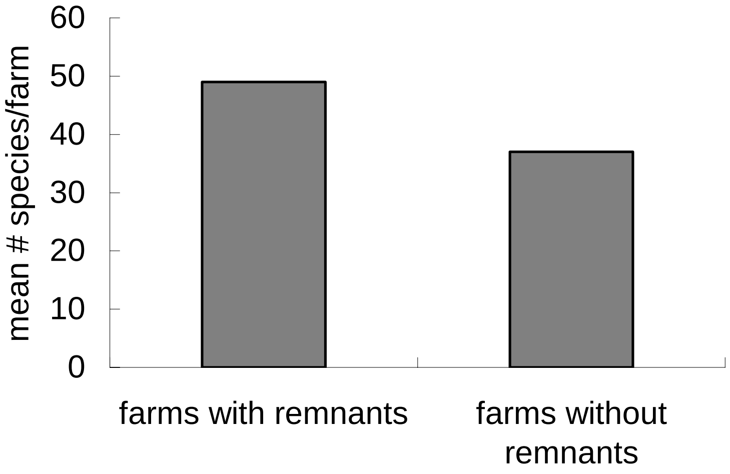


Figure 23. Influence of remnant vegetation patches on species richness of birds on 10 rice farms. n = 7 farms with remnants, n = 3 farms without remnants.

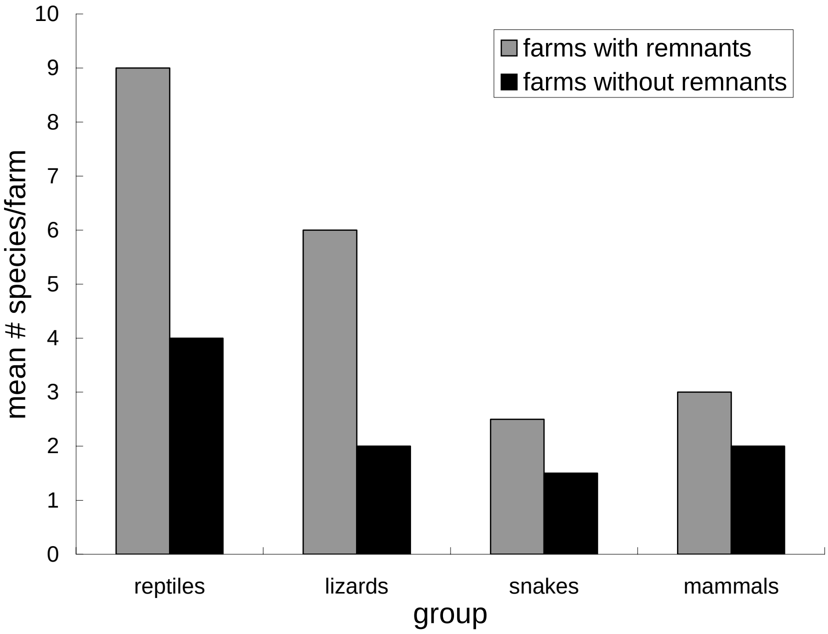


Figure 24. Influence of the presence of remnant vegetation patches on species richness of vertebrate groups on 10 rice farms. n = 7 farms with remnants, n = 3 farms without remnants.

Birds

Although farms with remnant vegetation patches held more bird species than farms without remnants (Fig. 23), the difference only approached significance when excluding Old Coree (F1,8 = 3.97; p = 0.087; means = 44.0 vs. 37.3 species) and was insignificant when including Old Coree (F1,9 = 1.80, p = 0.22; means = 49.3 vs. 37.3). However, more woodland specialist species were found on farms with remnant vegetation patches than on farms without remnants, when excluding Old Coree (F1,8 = 7.20, p = 0.031; means = 7.3 vs. 2.0 species), but the difference was insignificant when including Old Coree (F1,9 = 3.04, p = 0.12; means = 10.0 vs. 2.0).

Reptiles

Farms with remnant vegetation patches held more reptile species than farms without remnants (Fig. 24). This difference was significant when excluding Old Coree (F1,8 = 16.00, p = 0.005; means = 7.2 vs. 4.0), but only approached significance when including Old Coree (F1,9 = 3.80, p = 0.08; means = 8.6 vs. 4.0). Farms with remnant vegetation patches held more lizard species than farms without remnants when including Old Coree (F1,9 =6.86, p = 0.031; means = 5.7 vs. 1.7 species) or when excluding Old Coree (F1,8 =18.72, p = 0.003; means = 4.8 vs. 1.7 species). The number of snake species did not differ significantly between farms with remnant vegetation patches and farms without remnants when including Old Coree (F1,9 = 0.60, p = 0.463, means = 2.4 vs. 1.7), and without Old Coree (F1,8 = 0.26, p = 0.626; means = 1.8 vs. 1.7). Turtle species were too few (n=2) and too under-represented on farms to facilitate comparisons between farms with and without remnant vegetation patches.

Mammals

Farms with remnant vegetation did not hold significantly more mammal species than farms without remnants when including Old Coree (F1,9 = 1.40, p = 0.271; means = 6.7 vs. 4.3) and without Old Coree (F1,8 = 1.13, p = 0.323; means = 5.7 vs. 4.3) (Fig. 24). When considering only native mammals, farms with remnant vegetation patches did not hold significantly more mammal species than farms without remnants when including Old Coree (F1,9 = 1.37, p = 0.275; means = 3.3 vs. 1.7) and without Old Coree (F1,8 = 1.58, p = 0.250; means = 2.5 vs. 1.7). Considering exotic/feral mammals only, farms with remnant vegetation patches did not hold significantly more mammal species than farms without remnants when including Old Coree (F1,9 = 0.94, p = 0.361; means = 3.4 vs. 2.7) and when excluding Old Coree (F1,8 = 0.47, p = 0.516; means = 3.2 vs. 2.7).

Frogs

Farms with remnant vegetation patches held more frog species than farms without remnants when including Old Coree (F1,9 = 6.00, p = 0.040; means = 4.4 vs. 3.0) and without Old Coree (F1,8 = 6.73, p = 0.038; means = 4.2 vs. 3.0).

Habitat Utilisation Within Farms: Pitfall Trapping

Because pitfall trapping, when conducted on different farms simultaneously, is a relatively unbiased technique for detecting small animals, and because we employed pitfall trap-lines in different broad habitats, we were able to determine rough habitat utilisation within farms for several species. We divided trap-lines into different habitat ‘types’, and herein make paired comparisons of trap-line captures between: vegetation remnants and non-remnants, wet and dry habitats, rice vs. dry crop habitats, and dam vs. no dam habitats. Table 9 lists the number of trap nights for each trap-line. Because some trap-lines were open for slightly different amounts of time (Table 9), we corrected for effort by dividing the number of captures by the number of trap nights to facilitate statistical comparisons.

Table 9. Sampling effort of pitfall trap-lines on 10 rice farms, expressed as number of trap nights. Buckets were employed to capture small reptiles, small mammals, and frogs, while funnel traps were used to capture snakes and larger lizards. Only a few trap-lines were run in April. Hence, in some analyses we excluded the April data.

Trap-line	Buckets			Funnel traps		
	w/April	April	w/o April	w/April	April	w/o April
1	636	228	408	106	38	68
2	600	228	372	100	38	62
3	454	84	370	76	14	62
4	456	84	372	76	14	62
5	350	84	266	60	14	46
6	350	84	266	60	14	46
7	264	0	264	44	0	44
8	336	0	336	56	0	56
9	336	0	336	56	0	56
10	252	0	252	42	0	42
11	336	0	336	56	0	56
12	276	0	276	46	0	46
13	274	0	274	46	0	46
14	276	0	276	46	0	46
15	360	48	312	60	8	52
16	324	24	300	54	4	50
17	288	0	288	48	0	48
18	288	0	288	48	0	48
Total	6456	864	5592	1080	144	936

We conducted simultaneous pitfall trapping on the 10 farms during six 2-3 week periods in 2001-2002. This included 7532 trap nights (6456 buckets, 1076 funnel traps). A total of 9088 vertebrates were captured in pitfall traps during the study. Of the vertebrate groups captured in pitfall lines, frogs were the most frequently captured vertebrate (8691 = 97%), followed by lizards (127 = 1.5%), mammals (67 = 1%), snakes (37 = 0.5%), and turtles (1 = negligible %).

Capture Rates in Remnants vs. 'Non Remnants'

Data were sufficient to compare capture rates of several species and groups between trap-lines in remnant vegetation patches and trap-lines not associated with vegetation remnants. Reptiles, lizards, skinks, and snakes were captured more frequently in trap-lines in vegetation remnants than in trap-lines in other habitats, although the difference was not significant for snakes or for all reptiles (Table 10; Fig. 25). No significant difference in abundance of frogs and the house mouse existed between remnant trap-lines and non-remnant trap-lines (Table 10; Fig. 25). Three lizard species (wall lizard, olive legless lizard, and tessellated gecko) and red-bellied blacksnakes were captured only in trap-lines in remnant vegetation patches (Table 10).

Table 10. Comparison of capture rates between trap-lines among remnant vegetation patches (trap-lines 1,2,6,8,11,12,13,14,15,16) and trap-lines not associated with vegetation remnants (trap-lines 3,4,5,7,9,10,17,18). Capture rates are number of animals per trap night. Sample sizes are in parentheses. NS = non significant, ** = denotes significance at $p < 0.01$; *** denotes significance at $p < 0.001$.

Species	Remnants mean ± SD	Non-remnants mean ± SD	Statistics (single factor ANOVA)
spotted grass frog	0.58 ± 0.67 (1741)	0.93 ± 0.49 (2295)	F1,17 = 1.56, p = 0.230; NS
barking marsh frog	0.05 ± 0.07 (138)	0.13 ± 0.12 (314)	F1,17 = 3.34, p = 0.09; NS
plains froglet	0.02 ± 0.06 (65)	0.02 ± 0.02 (38)	F1,17 = 0.035, p = 0.854; NS
all frogs	0.64 ± 0.75 (1945)	1.08 ± 0.54 (2648)	F1,17 = 1.87, p = 0.190; NS
eastern brownsnake	0.01 ± 0.02 (6)	0.02 ± 0.02 (10)	F1,17 = 1.76, p = 0.204; NS
red-bellied blacksnake	0.02 ± 0.04 (10)	0 (0)	n/a
all snakes	0.05 ± 0.04 (24)	0.02 ± 0.02 (11)	F1,17 = 1.44, p = 0.247; NS
boulenger’s skink	0.03 ± 0.02 (63)	0.002 ± 0.003 (5)	F1,17 = 11.29, p = 0.004; **
wall lizard	0.003 ± 0.004 (8)	0 (0)	n/a
all skinks	0.03 ± 0.02 (94)	0.003 ± 0.004 (8)	F1,17 = 14.90, p = 0.001; ***
tessellated gecko	0.002 ± 0.004 (9)	0 (0)	F1,17 = 2.16, p = 0.161; NS
olive legless lizard	0.001 ± 0.005 (6)	0 (0)	n/a
all lizards	0.04 ± 0.02 (112)	0.003 ± 0.004 (8)	F1,17 = 24.07, p = 0.0001; ***
all reptiles	0.11 ± 0.17 (135)	0.006 ± 0.005 (19)	F1,17 = 2.98, p = 0.103; NS
house mouse	0.006 ± 0.01 (17)	0.007 ± 0.007 (16)	F1,17 = 0.070, p = 0.794; NS

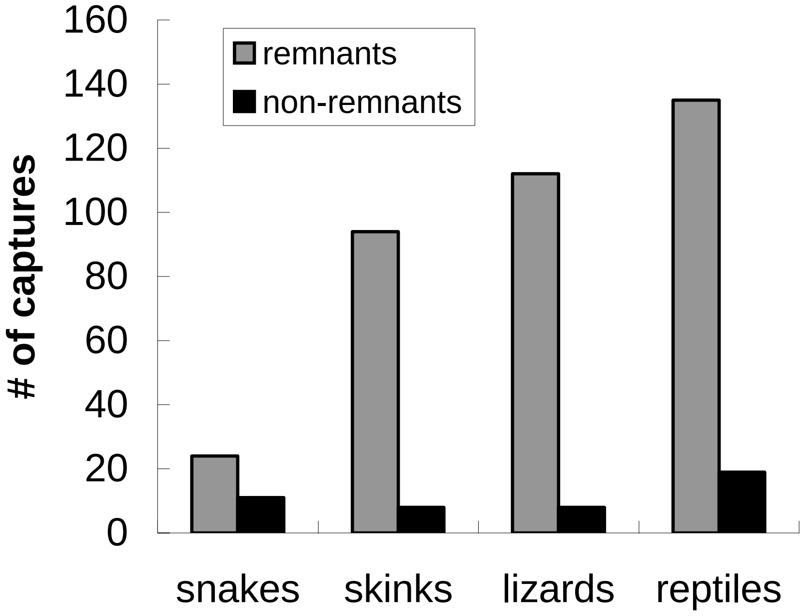


Figure 25. Influence of the presence of remnant vegetation patches on abundance of snakes, skinks, lizards, and all reptiles, given equal survey effort. Captures are from pitfall trap-lines. April captures are excluded.

Capture Rates in Wet vs. Dry Habitats

Data were sufficient to compare capture rates of several species and groups between 'wet' and 'dry' habitats (Table 11). Analyses indicate that all frogs combined, spotted grass frogs, and barking marsh frogs, were more abundant in rice crops than in dry crops, although the value for barking marsh frogs only approaches significance (Table 11; Fig. 26).

Table 11. Comparison of capture rates between trap-lines in ‘wet’ habitats (trap-lines 3, 5, 7, 8, 9, 13, 16, 17) and trap-lines in ‘dry’ habitats (trap-lines 1, 2, 4, 6, 10, 11, 12, 14, 15, 18). Capture rates are number of animals per trap night. Sample sizes are in parentheses. NS = non significant, ** denotes significance at p < 0.01, app = approaching significance.

Species	‘wet’ mean ± SD	‘dry’ mean ± SD	Statistics (single factor ANOVA)
spotted grass frog	1.02 ± 0.58 (3133)	0.29 ± 0.33 (903)	F1,17 = 9.21, p = 0.008; **
barking marsh frog	0.12 ± 0.11 (376)	0.03 ± 0.05 (76)	F1,17 = 4.23, p = 0.056; app
plains froglet	0.02 ± 0.06 (78)	0.01 ± 0.02 (27)	F1,17 = 0.23, p = 0.636; NS
all frogs	1.16 ± 0.65 (3585)	0.33 ± 0.35 (1008)	F1,17 = 9.42, p = 0.007; **
eastern brownsnake	0.02 ± 0.02 (10)	0.01 ± 0.02 (6)	F1,17 = 0.71, p = 0.412; NS
red-bellied blacksnake	0.02 ± 0.04 (6)	0.003 ± 0.008 (4)	F1,17 = 0.77, p = 0.392; NS
all snakes	0.04 ± 0.03 (17)	0.03 ± 0.04 (18)	F1,17 = 0.02, p = 0.887; NS
boulenger’s skink	0.02 ± 0.2 (35)	0.02 ± 0.01 (53)	F1,17 = 0.01, p = 0.927; NS
wall lizard	0.002 ± 0.004 (3)	0.001 ± 0.001 (5)	F1,17 = 1.70, p = 0.211; NS
all skinks	0.02 ± 0.02 (42)	0.02 ± 0.01 (60)	F1,17 = 0.08, p = 0.781; NS
all lizards	0.02 ± 0.02 (43)	0.02 ± 0.02 (76)	F1,17 = 0.08, p = 0.782; NS
all reptiles	0.04 ± 0.08 (60)	0.09 ± 0.20 (94)	F1,17 = 0.55, p = 0.467; NS
house mouse	0.008 ± 0.01 (15)	0.004 ± 0.005 (18)	F1,17 = 0.71, p = 0.411; NS

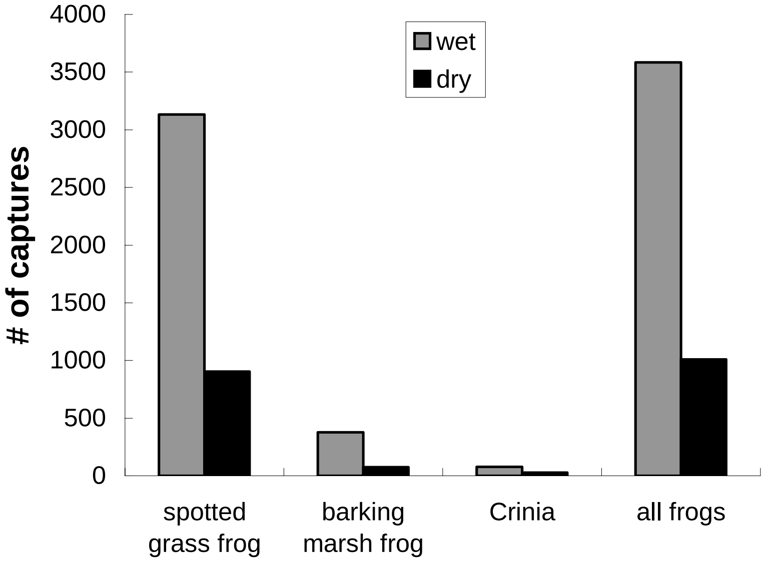


Figure 26. Influence of wet vs. dry habitats on abundance of frogs, given equal survey effort. Wet habitats included rice bays, dams, and swamps, while dry habitats were canola, fallow fields, and dry black box depressions. Captures are from pitfall trap-lines.

Capture Rates in Rice vs. Dry Crops

The above analysis of wet vs. dry habitats includes wet habitats other than rice, including farm dams and ephemeral swamps, and dry habitats that are not crops, such as grasslands and remnants. Thus, to understand the how rice compares to dry crops in harbouring biodiversity, we analysed rice vs. dry crops (wheat, canola, fallow). Data were sufficient to compare capture rates of the several species and groups between rice and dry crops (Table 12).

Analyses indicate that only spotted grass frogs were more abundant in rice crops than in dry crops (Table 12). However, mean capture rates for other frogs (and all frogs) were invariably higher in rice than in dry crops, despite the lack of statistical significance (Table 12).

Table 12. Comparison of capture rates of pitfall trap-lines between rice habitat (trap-lines 7, 8, 9, 13, 16, 17) and dry crop habitat (wheat, canola, fallow; fences 2, 4, 5, 18). Capture rates are number of animals per trap night. Sample sizes are in parentheses. NS = non significant, * denotes significance at $p < 0.05$.

Species	Rice mean ± SD	Dry crops mean ± SD	Statistics (single factor ANOVA)
spotted grass frog	1.23 ± 0.41 (2224)	0.60 ± 0.24 (995)	F1,8 = 5.86, p = 0.046; *
barking marsh frog	0.16 ± 0.13 (288)	0.03 ± 0.04 (59)	F1,8 = 2.93, p = 0.131; NS
plains froglet	0.040 ± 0.08 (72)	0.01 ± 0.01 (10)	F1,8 = 0.44, p = 0.528; NS
all frogs	1.43 ± 0.46 (2584)	0.89 ± 0.57 (1064)	F1,8 = 2.74, p = 0.137; NS
eastern brownsnake	0.02 ± 0.02 (7)	0.02 ± 0.03 (4)	F1,8 = 0.21, p = 0.660; NS
boulenger’s skink	0.02 ± 0.03 (35)	0.01 ± 0.004 (11)	F1,8 = 0.23, p = 0.644; NS
all skinks	0.07 ± 0.13 (40)	0.01 ± 0.005 (13)	F1,8 = 0.57, p = 0.474; NS
all lizards	0.03 ± 0.03 (41)	0.02 ± 0.01 (18)	F1,8 = 0.25, p = 0.630; NS
all reptiles	0.03 ± 0.02 (54)	0.01 ± 0.007 (24)	F1,8 = 0.55, p = 0.489; NS
house mouse	0.004 ± 0.004 (7)	0.003 ± 0.005 (8)	F1,8 = 0.02, p = 0.888; NS

Spotted Grass Frogs vs. Barking Marsh Frogs

During April 2001 heavy rainfall resulted in a large number of captures of two related frog species, the spotted grass frog and the barking marsh frog, in four trap-lines on two farms (Marsden, Stephenson). Because one trap-line on each farm was next to a farm dam, and one was not, we had the opportunity to compare the abundance of the two species between sites with dams and sites without dams.

Of 1574 captures of the two species, 889 were barking marsh frogs (BMF’s) and 685 were spotted marsh frogs (SMF’s) (Fig. 27). Species was not independent of habitat type (dam vs. other, $\chi 2 = 235.17$, df = 1, $p < 0.001$; Fig. 27). BMF’s were more likely to be associated with farm dams than SMF’s (Fig. 27).

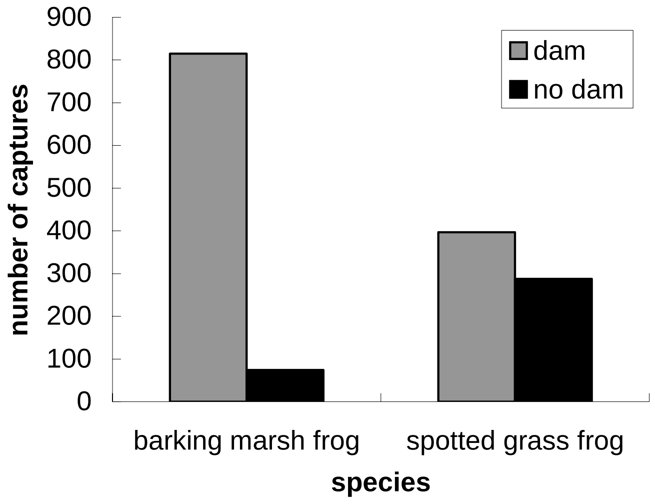


Figure 27. Association between habitat type (dam vs. no dam) and abundance of two species of frogs on two rice farms.

Spotted Grass Frog Production in Rice Bays

In April 2001 a fortuitous situation allowed us to crudely estimate the number of spotted grass frogs produced in rice bays in the Riverina each year. March and early April 2001 were very dry, with little or no measurable rain at the study farms. During this time the rice crops were harvested and the bays drained, leaving a small amount of water in the bays, along with rice crop stubble. Apparently, during this time, many or all of the spotted grass frog tadpoles had transformed into frogs, and were waiting for significant rainfall to disperse from the bays.

On the day and night of 21 April, the farms received heavy rainfall (27 mm at Old Coree, Russell Ford, Rice Research, unpubl. data). During this time we had begun to install pitfall trap-lines; trap-lines were functional on three farms. The next day one of us (SD) discovered great magnitudes of young frogs (mostly spotted grass frogs) in the pitfall buckets, particularly along one trap-line at Old Coree. A total of 1631, > 95% yearling, spotted grass frogs (Limnodynastes tasmaniensis) were captured in the trap-line buckets at the Old Coree trap-line during the night of 21 April. Only two of these frogs were in buckets on the remnant side of the rice bay side of the trap-line, indicating that the frogs had dispersed from the rice bays.

This was the only trap-line at Old Coree that was adjacent to rice bays (approx. 10 m away from, and parallel to the closest bay). On the other side of the trap-line was riparian remnant vegetation (grazed, open eucalypt woodland with a few saltbush along a large 20 m wide slow flowing Billabong Creek). We hypothesized that this was a major dispersal event for the frogs for the year, though it was likely not the only dispersal event, as spotted grass frogs have an extended breeding season (Barker et al., 1995). Under the assumption that the frog production of these rice bays was typical or average, we estimated the number of frogs produced annually in NSW Riverina rice bays. This required three steps: estimating the number of frogs dispersing from the bays, correcting frog numbers for one hectare of rice, and extrapolating to the number of hectares of rice produced in a typical year.

First, we assumed that frog movements from the bays was equal in all directions, and therefore the proportion of our captures to the total number of frogs dispersing from the bays was equivalent to the proportion of the length of our trap-line to the length of a trap-line that would completely surround the bays:

$$\frac{\text{\# of captures}}{\text{\# of frogs dispersing}} = \frac{\text{length of trap-line}}{\text{length of trap-line if it completely surrounded the bays}}$$

Using this equation we arrived at a number of 52,192 frogs dispersing from the bays. We then calculated how many frogs would be produced in one hectare of rice, given that the bays covered 15 hectares (R. Ford, Rice Research, pers. comm.). We arrived at a figure of 3,479 frogs produced per hectare. Finally, we extrapolated to the number of hectares of rice produced in a typical year on rice farms in the NSW Riverina. We were informed that 150,000 hectares of rice is produced annually in the Riverina (M. Linnegar, Ricegrowers' Association, pers. comm.). Multiplying our 3,479 frogs/hectare by 150,000 hectares yields a very crude estimate of 521,850,000 spotted grass frogs produced from rice bays in the NSW Riverina annually. If there were more dispersal events for the species during the year with equivalent numbers, our estimates may be conservative, and numbers could be much higher (e.g., 2 dispersal events = 1,043,700,000 frogs; 5 dispersal events = 2,609,250,000). We acknowledge that we do not have replication and therefore, an error estimate for this finding.

Bird Diversity and Flooded Rice Bays

The diversity of birds utilising rice bays increased following the flooding of rice bays (Fig. 28; Fig. 29). Both species richness and total abundance of birds counted increased from September (when bays were dry) to October (after bays were flooded). Both species richness and total abundance decreased after the rice bays were drained (Fig. 28, 29). Although the influence of month on species richness and total abundance was not statistically significant (richness, F4,37 = 1.59, p = 0.200; abundance, F4,37 = 1.31, p = 0.285), richness and abundance were significantly higher when bays were flooded than when bays were not flooded (richness, F1,37 = 6.58, p = 0.015; means = 11.1 vs. 6.7 species/count/farm; abundance, F1,37 = 5.5, p = 0.024; means = 136.7 vs. 34.6 birds/count/farm). Birds counted when bays were not flooded were mainly non-aquatic species, especially in September (e.g., ravens, magpie larks, starlings, zebra finches, lapwings, galahs, parrots, swallows, raptors). However, some aquatic species did utilise drained rice bays in April (e.g., white-faced herons, white-necked heron, straw-necked ibis, white ibis).

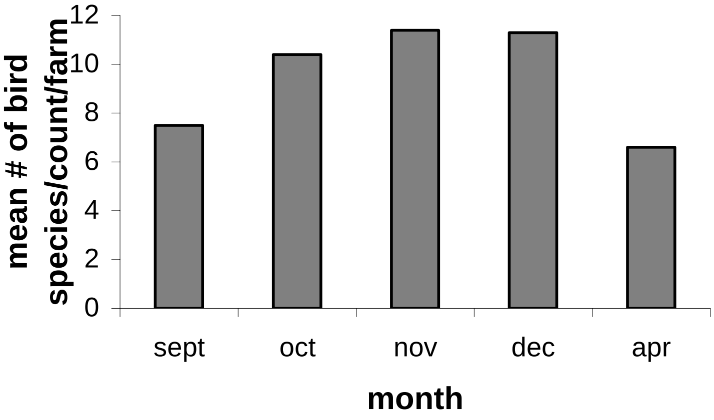


Figure 28. Changes in species richness of birds utilising rice bays, showing increase in richness associated with flooding of the rice bays in Sept/Oct, and similar decrease in richness associated with draining of the rice bays in March. Numbers are means of monthly counts on each of 10 farms.

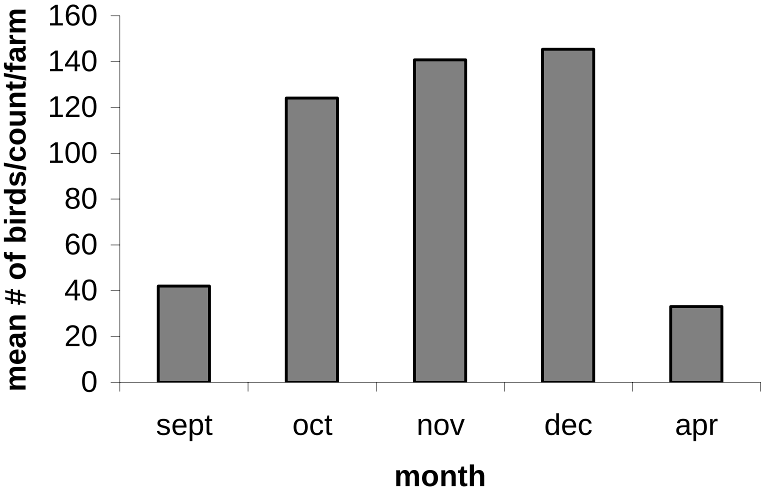


Figure 29. Changes in abundance of birds utilising rice bays, showing increase in abundance associated with flooding of the rice bays in Sept/Oct, and similar decrease in abundance associated with draining of the rice bays in March. Numbers are means of monthly counts on each of 10 farms.

II. The Intensive Studies

A. Bird Diversity and Characteristics of Remnant Vegetation Patches

Bird Diversity and Breakdown

Remnant Vegetation Study Sites

Twenty-three sites were surveyed for birds between 2 October 2002 and 7 December 2002. Each site was visited on eight occasions, totaling 184 surveys across all sites. Surveys had a duration of 15 minutes, which gave a survey effort of 120 minutes at each site and a total of 2760 minutes across all sites.

Diversity of Birds Utilising Remnant Vegetation Study Sites

Within or in the immediate vicinity of the Coleambally Irrigation Area (CIA) a total of 115 bird species including 93 land bird species were recorded during the ‘birds and remnants study (Appendix 3). Seventy-two species were recorded utilising the remnant vegetation study sites, representing 58 genera and 28 families (Appendix 3). Using the categories identified by Reid (1999), 11.1% of these species were DECREASERS, 31.9% INCREASERS, 40.3% WOODLAND, and 16.7% OTHER. Of the twenty-one most frequently encountered species that were recorded across most sites, 71.4% were INCREASERS, 23.8% were WOODLAND, 4.8% were OTHER and 0.0% were DECREASERS (Table 13). Two species of birds, the superb parrot (Polytelis swainsonii) and grey-crowned babbler (Potmatostomus temporalis), which are listed as being nationally vulnerable and near threatened species, respectively, were recorded using remnants in the study.

Using the categories identified by Reid (1999), eight of the 20 DECREASER species and 23 of the 36 INCREASER species were recorded during the present study (Appendix 2). Of the remainder of species recorded, 29 were WOODLAND and 12 were OTHER species (Seddon et al., 2001).

Table 13. The 21 most frequently encountered species across all sites, and the species groups that they occupied. Species classified as INCREASERS (I) follows Reid (1999), those classified as WOODLAND (W) or OTHER (O) follows Seddon *et al*. (2001) (see Methods).

Species	% sites occupied	species group
magpie-lark	100	I
Australian raven	96	I
crested pigeon	96	I
Australian magpie	91	I
common starling	91	I
Galah	91	I
noisy miner	91	I
red-rumped parrot	87	I
striated pardalote	87	W
Cockatiel	74	I
black-faced cuckoo-shrike	70	I
willie wagtail	70	I
blue bonnet	61	I
grey butcherbird	61	W

pied butcherbird	48	I
white-faced heron	48	O
white-plumed honeyeater	48	I
Weebill	43	W
superb parrot	39	W
white-winged chough	39	I
yellow thornbill	39	W

Justification of Survey Replication

Examination species accumulation curves and the use of a negative exponential model to estimate the expected number of species at each site indicated that eight replications was sufficient to estimate total species richness (Table 14). On average, across all sites, 90.5% of the total species that had been observed at that site had been located after six replications (Table 14). The observed species richness at two of the 23 sites (84 and 194) fell outside the approximate 95% confidence intervals of the expected values (Table 14).

Table 14. The observed and expected species richness across sites, calculated by a negative exponential model, including the approximate 95% confidence intervals of the expected value. % = the percentage of species recorded after six replications compared to that after eight replications.

			Approximate 95% Confidence intervals		
Site no	Observed	Expected	Min	Max	%
12	25	30.79	24.10	37.49	88.0
15	16	15.10	12.90	17.30	87.5
42	20	19.24	17.89	20.59	90.0
49	13	13.24	12.80	13.68	92.3
71	23	20.67	17.16	24.18	82.6
84	19	21.52	19.42	23.61	94.7
89	17	16.95	14.68	19.22	88.2
95	23	22.47	20.10	24.84	91.3
136	18	18.27	16.90	19.64	94.4
165	19	18.00	17.10	18.91	94.7
168	23	23.71	21.96	25.46	91.3
179	24	24.58	21.69	27.48	91.7
184	21	22.13	18.94	25.31	100.0
192	19	19.97	17.54	22.40	94.7
194	32	37.89	34.36	41.43	87.5
198	15	16.39	12.80	19.97	93.3
201	19	41.65	3.92	79.38	73.7
205	21	23.89	19.92	27.86	90.5
206	33	33.62	30.45	36.78	90.9
207	19	19.88	17.58	22.17	89.5
208	30	30.34	27.92	32.77	93.3

209	19	20.49	16.92	24.08	89.5
210	25	26.66	24.67	28.64	92.0

Summary of Landscape and Habitat Variables

Landscape and habitat variables across the four patch size classes (very small, small, medium, large) is listed in Table 15. Patch area (log) was not related to habitat complexity (F1, 21 =1.23; p= 0.274; r2= 0.05) or total shrub cover (F1, 21= 0.703; p= 0.411; r2= 0.03).

Table 15. Summary statistics for landscape and habitat variables across site classes. N = the number of remnants within each site class. SE = standard error. See Table 2 for an explanation of abbreviations.

		Very small		**Small**		**Medium**		**Large**	
		<2ha	*n=4*	*2-10ha*	*n=7*	*10-50ha*	*n=7*	*>50ha*	*n=5*
Landscape variables		**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
1	DTNP >10ha (m)	587.50	210.04	757.86	362.51	991.43	839.05	129.60	24.21
2	DTNP >100ha (m)	4511.25	1223.04	2628.57	838.10	3822.14	1552.75	1562.20	1033.12
3	DTNPSV >10ha (m)	1214.25	345.91	947.28	529.97	1326.71	950.46	237.20	96.61
4	2km NOP	3.00	1.08	5.00	1.31	5.14	1.22	6.80	0.66
Habitat variables									
1	P area (ha)	1.45	0.18	5.07	1.17	28.91	4.60	105.41	24.16
2	P shape	21.61	1.92	32.22	4.20	66.79	10.76	124.71	25.39
3	CCO %	36.25	8.98	43.57	6.48	35.71	6.81	29.66	3.95
4	CCTS (2-4m) %	2.50	2.50	3.57	2.46	2.61	1.45	6.66	1.16
5	CCS (0.5-2m) %	10.00	6.77	6.43	4.98	1.90	0.92	20.00	4.18
6	DABH	3.50	1.85	2.71	0.58	3.14	0.46	4.40	0.84
7	Dead T	3.25	0.48	1.79	1.15	1.14	0.73	1.15	0.26
8	Hollows	27.00	11.01	16.07	3.39	20.66	4.80	26.06	4.68
9	HG cov %	28.75	23.84	29.29	8.20	27.14	5.55	16.46	0.93
10	BG cov %	41.25	14.20	30.36	9.28	26.31	5.40	49.00	10.16
11	BL cov %	30.00	10.21	40.36	9.29	46.54	7.76	34.54	5.06
12	Logs (m3)	0.32	0.07	0.35	0.10	0.57	0.13	0.83	0.16
13	MD	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.03
14	Water	0.75	0.25	0.71	0.16	0.86	0.14	0.60	0.09
15	Canopy (m)	12.75	0.85	13.64	1.66	14.10	0.87	15.00	2.91
16	Regen	37.50	37.17	7.79	10.11	8.30	7.57	17.68	2.27

17	Dieback	1.75	0.25	1.00	0.12	1.71	0.42	1.40	0.19
18	Fire	0.50	0.29	0.29	0.25	0.29	0.29	0.40	0.40
19	Stems	18.50	4.79	18.21	4.04	10.83	1.98	9.60	1.20
20	DABHR	6.25	1.38	6.57	0.87	4.29	0.99	6.40	0.81

A principal component analysis (PCA) ordination conducted on a subset of landscape and habitat variables (Table 16) indicated that Site 210 was distinct from all other sites (Fig. 30). Sites 12, 15, 71, 179, and 192 were also distinct from the remaining sites, which were more tightly clustered (Fig. 30). PC one, two and three explained 22.2, 16.4 and 14.3% of the variation in the subset of landscape and habitat variables respectively (Fig. 30; Table 16). Negative influences on PC 1 consisted of DTNP >10ha, DTNP >100ha, DTNPSV >10ha and HG cov (Table 16). Positive influences included 2km NOP, P area, P shape, CCTS, CCS and Regen (Table 16). Negative influences on PC 2 consisted of DTNP >10ha, P shape, CCTS, CCS, DABH, Dead t, Hollows and Regen (Table 16). Positive influences included 2km NOP, CCO, BL cov, Logs and Stems (Table 16).

Table 16. Coefficients in the linear combinations of variables making up the principal components within the Principal Components Analysis. The three PC's explained 52.9% of the variation in the data. See Table 2 for an explanation of abbreviations. Log and square root transformations to normalise data are denoted by a and b, respectively.

Variable	**PC1**	**PC2**	**PC3**
DTNP >10ha [a]	-0.373	-0.205	0.069
DTNP >100ha [a]	-0.371	-0.077	0.110
DTNPSV >10ha [a]	-0.385	-0.090	0.168
2km NOP [a]	0.322	0.259	-0.194
P area [a]	0.244	-0.189	-0.030
P shape [a]	0.215	-0.312	0.145
CCO	0.096	0.252	0.341
CCTS (2-4m)	0.249	-0.282	0.373
CCS (0.5-2m)	0.215	-0.323	0.345
DABH	0.141	-0.209	-0.250
Dead t [a]	-0.154	-0.205	0.087
Hollows [b]	0.023	-0.245	-0.311
HG cov [a]	-0.212	-0.060	0.094
BL cov [a]	0.189	0.348	0.309
Logs [b]	0.186	0.228	-0.056
Regen [b]	0.244	-0.265	0.269
Stems	-0.097	0.294	0.324
DABHR	0.139	-0.132	-0.262

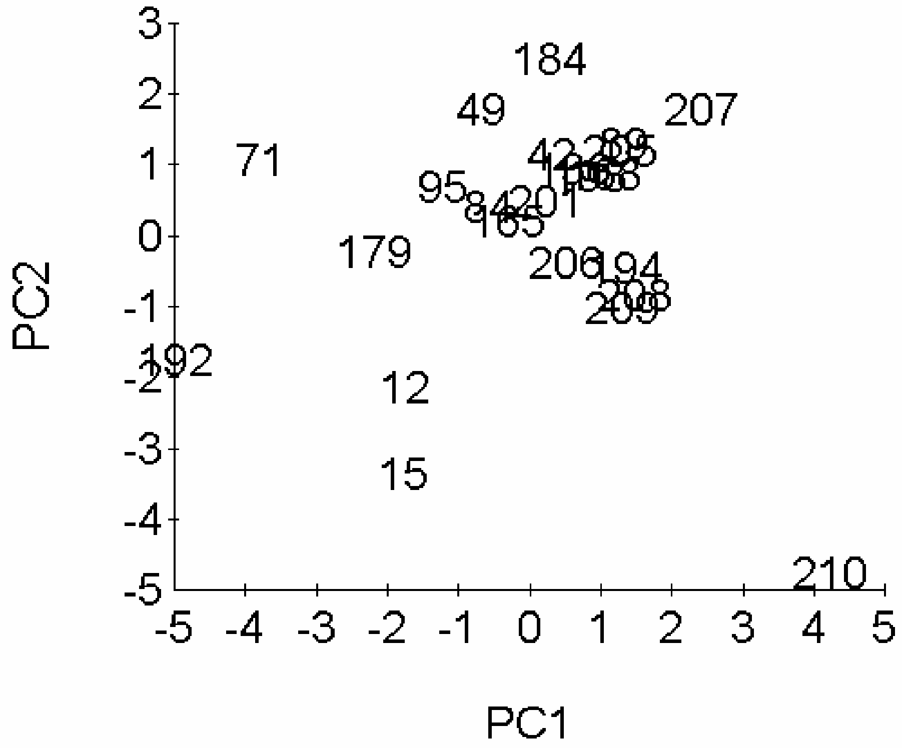


Figure 30. Two-dimensional principal component analysis (PCA) ordination of the sub-set of habitat and landscape variables across sites (from Table 16). PC 1 and PC 2 collectively accounted for 38.6% of the total sample variation.

Bird Assemblage Similarities Across Sites

Clustering and Non-metric Multidimensional Scaling

Based on mean logged bird abundance counts, at 50% group-average clustering from Bray-Curtis similarities, three distinct groups were evident (Fig. 31). These sets of groupings were superimposed onto the nMDS ordination and it was clear that the two techniques were in agreement in assigning groups at that level of similarity (Fig. 31; Fig. 32 a, b). Group 1 consisted of Site 210, a large site that acted as the pseudocontrol for the present study (Fig. 31; Fig. 32 a, b; see Methods for description of pseudocontrol site). Site 210 shared less than 30% similarity in mean bird abundance when compared to any other site based on group-average clustering from Bray-Curtis similarities (Fig. 31). Group 2 consisted of a mixture of one small (site 71) and three very small sites (sites 12, 15 and 42) (Fig. 31, Fig. 32 a, b). Group 3 was composed of the remaining sites that represented all size classes (Fig. 31, Fig. 32 a, b). Based on group-average clustering of Bray-Curtis similarities formed at a 55 to 60% level of similarity on logged mean abundance counts, group three split into a number of sub-groups (Fig. 31). The stress in two dimensions for the non-metric multidimensional scaling (nMDS) ordination was 0.14 (Fig. 32 a, b) whilst in three dimensions it was 0.08.

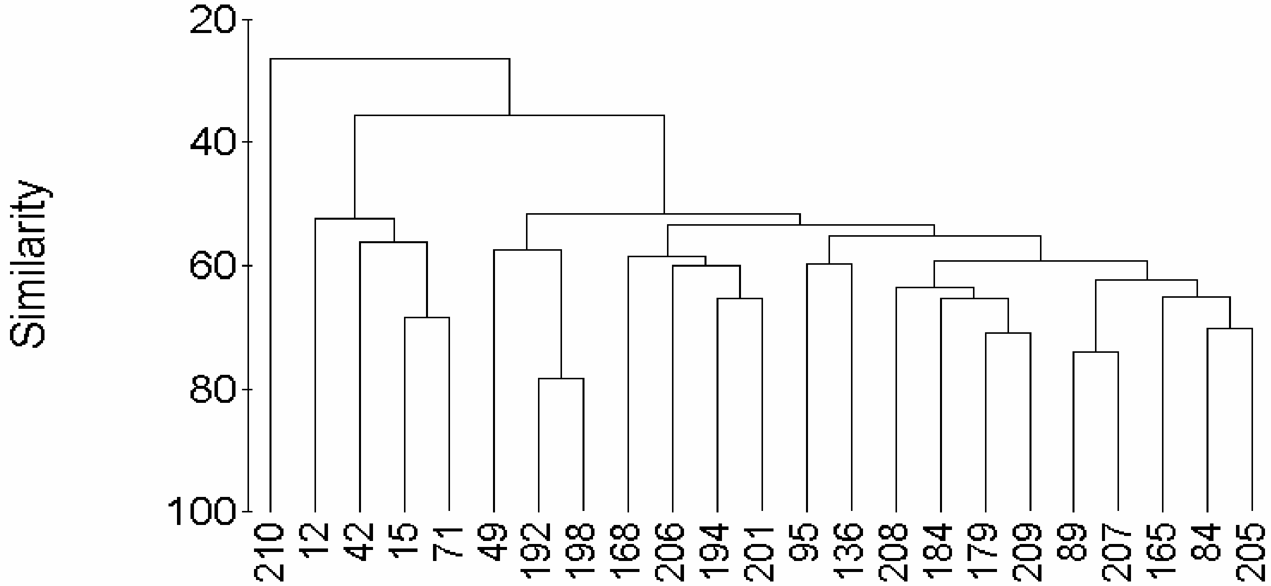
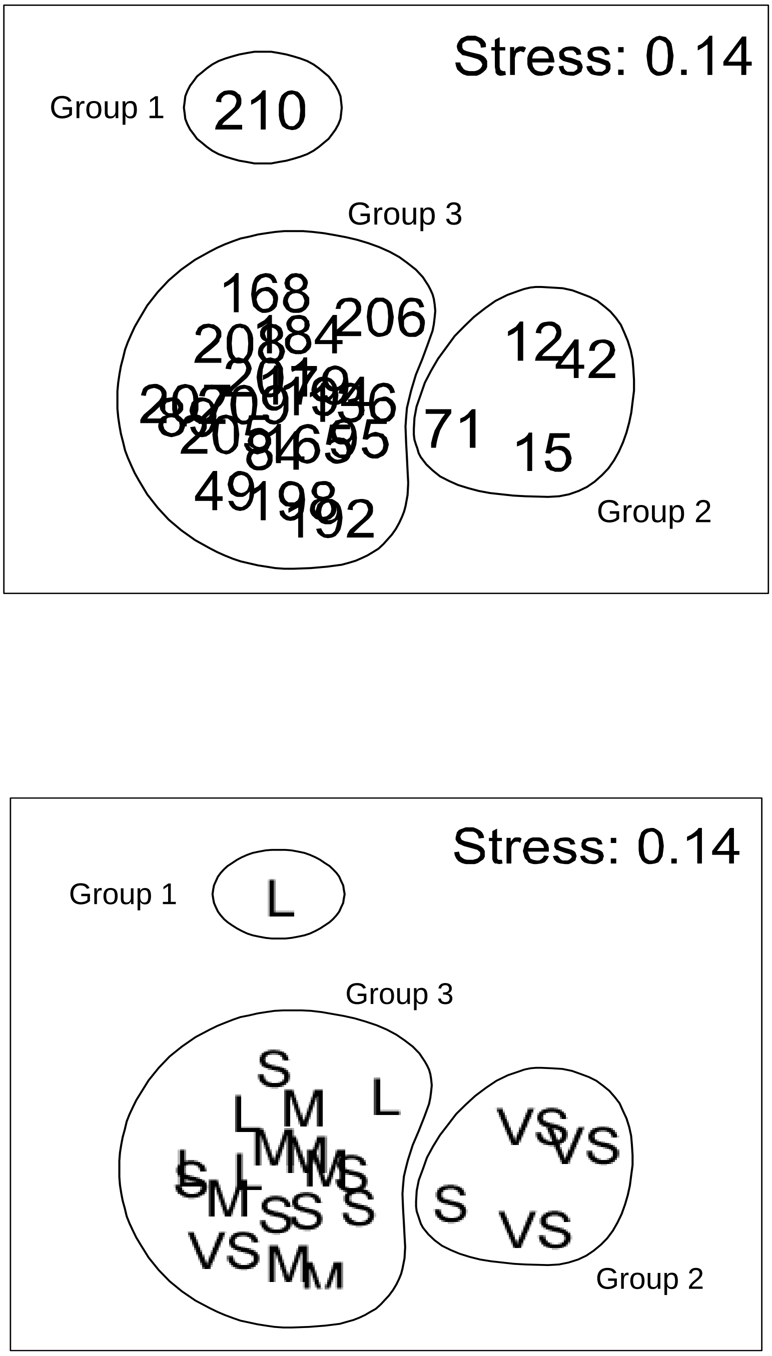


Figure 31. Dendrogram for hierarchical clustering for the 23 sites representing the group-average linking of Bray-Curtis similarities between sites calculated on logged bird mean abundance. Sites classes are: Very Small (<2ha) = sites 12, 15, 42 and 49; Small (2-<10ha) = sites 71, 84, 89, 95, 136, 165 and 168; Medium (10-50ha) = sites 179, 184, 192, 194, 198, 201 and 205; and Large (>50ha) = sites 206, 207, 208, 209 and 210.



(a)

(b)

Figure 32. Non-metric multidimensional scaling (*n*MDS) ordination of the logged bird mean abundance across (a) sites by (b) size class. Groups represent a group-average clustering from Bray-Curtis similarities at 50%. VS = very small; S = small; M = medium; L = large; see Methodology for a more detailed description.

Similarity Percentages Analysis

A similarity percentages (SIMPER) analysis was conducted to assess the average similarity of bird assemblages within site classes and the average dissimilarity between site classes based on mean abundance. This analysis also allowed for the determination of which species were contributing the most to the observed differences in bird assemblages between size classes. Average similarities were 38.3, 51.7, 57.2 and 45.8% within the very small, small, medium and large size classes, respectively. The average dissimilarity between size classes was 58.2% for very small vs. small, 61.7% for very small vs. medium, 68.4% for very small vs. large, 45.0% for small vs. medium, 52.5% for small vs. large, and 49.1% for medium vs. large.

The three species contributing most to the dissimilarity between each pair of site classes and the % dissimilarity they contribute are: (1) very small vs. small = house sparrow (9.04%), noisy miner (8.76%) and white-plumed honeyeater (5.44%); (2) very small vs. medium = house sparrow (9.91%), noisy miner (9.29%) and white-plumed honeyeater (5.65%); (3) very small vs. large = house sparrow (8.37%), noisy miner (7.05% and common starling (5.89%); (4) small vs. medium = red-rumped parrot (6.76%), white-winged chough (5.20%) and blue bonnet (4.64%); (5) small vs. large = white-winged chough (5.43%), common starling (5.20%) and red-rumped parrot (4.67%); (6) medium vs. large = white-winged chough (5.55%), red-rumped parrot (5.47%) and yellow thornbill (4.72%).

Analysis of Similarity

An analysis of similarity (ANOSIM) was conducted to detect statistically significant differences in bird assemblage composition based on logged mean abundance counts. A significant overall area effect on bird assemblage composition among size classes was found (Global R = 0.198; p = 0.013). However, after a Bonferroni correction was applied the difference was not significant (p = 0.833). Bird assemblage composition was significantly different between very small (n = 4) and small (n = 7) remnants (Global R = 0.354; p = 0.033); between very small and medium remnants (n = 7) (Global R = 0.585; p = 0.015); and between very small and large remnants (n = 5) (Global R = 0.431; p = 0.032). In contrast, no statistical differences existed between small and medium remnants (Global R = -0.049; p = 0.664), between small and large remnants (Global R = 0.047; p = 0.304), or between medium and large remnants (Global R = 0.100; p = 0.199).

Presence/Absence, Species Richness, Species Abundance/Density, and Composition Across Sites

Presence/Absence of Bird Species

Area-sensitive species (n=17) were restricted to medium and large size classes (Table 17). These species were comprised of eleven WOODLAND species, five DECLINER species, and one OTHER species. Site 210, the pseudocontrol site for the study, had four species unique to it alone.

Some habitat generalist species were restricted to the very small and small size classes (Table 17), while other generalist species were found across all, or three of the four size classes (Appendix 4). Of the species restricted to the small and very small size classes, four were classified as OTHER species, three were WOODLAND species, and two were INCREASER species.

Table 17. Generalist species that were restricted to very small or small remnants, and area-sensitive species found only in medium or large remnants.

Habitat generalists species		Area-sensitive species	
Very small <2ha (n=4)	Small 2-10 ha (n=7)	Medium 10-50 ha (n=7)	Large >50 ha (n=5)
White-necked Heron	Black -shouldered Kite	Pallid Cuckoo	Emu
Brown Goshawk	Yellow-throated Miner	Chestnut-rumped Thornbill	Australian Hobby
Collared Sparrowhawk	Singing Honeyeater	Buff-rumped Thornbill	Horsefield's Bronze-cuckoo
Crimson Rosella		Restless Flycatcher	Australian Owlet-nightjar
Budgerigar		Mistletoebird	Rainbow Bee-eater
Welcome Swallow			Western Gerygone
			Inland Thornbill
			Little Friarbird
			Black Honeyeater
			Jacky Winter
			Red-capped Robin
			Grey Fantail

Species Composition

For species richness, INCREASERS made-up the proportional bulk of species across sites, with the exception of the pseudocontrol site, 210 (Fig. 33). Only at sites 210, 208 and 206 did the proportion of INCREASERS drop below 50% (Fig. 33). The second most highly-represented group was the WOODLAND birds, followed by the OTHER birds group, and the DECREASERS group (Fig. 33).

INCREASERS were also the most dense/abundant group, followed by WOODLAND birds, DECREASERS, and OTHER birds (Fig. 34). Therefore, each INCREASER species on average, contributes a greater number of individuals to the assemblage when compared to other species groups. Only at Site 210 did the proportional abundance of INCREASERS drop below 65% (Fig. 34).

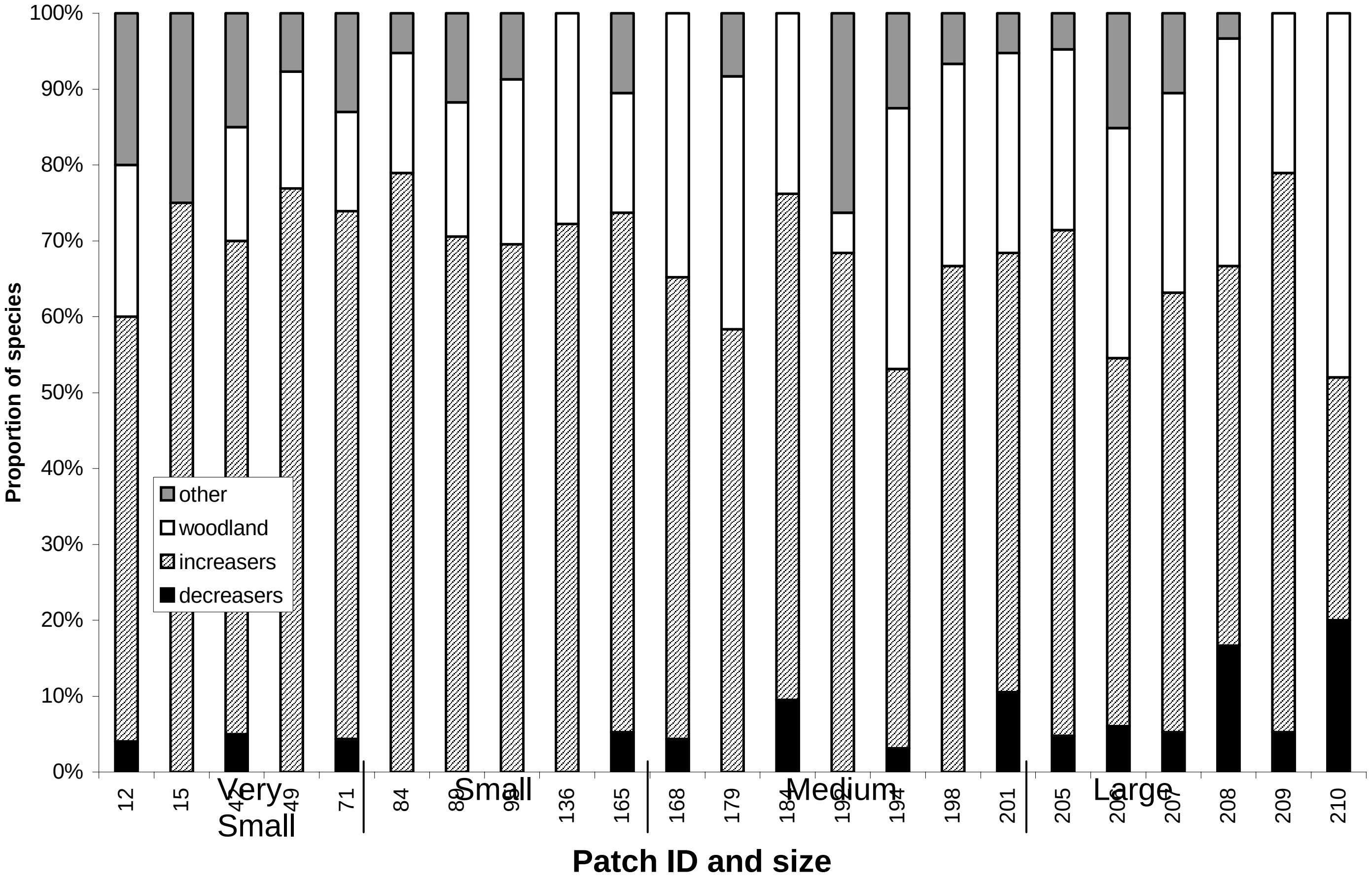


Figure 33. Proportion of different species groups across sites. Species classified as INCREASERS and DECREASERS follow Reid (1999), and species classified as OTHER and WOODLAND follow Seddon *et al*. (2001) (see Methods).

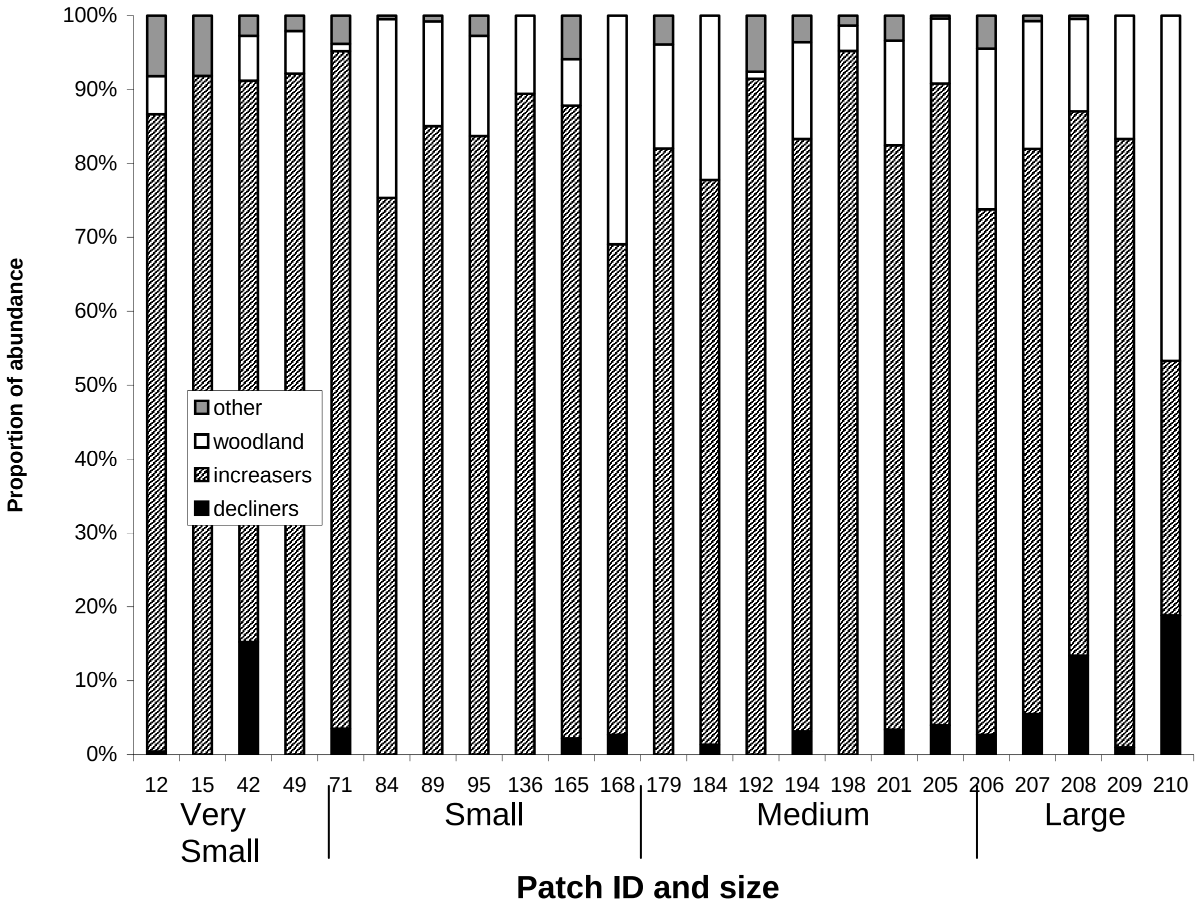


Figure 34. Proportion of abundances of different species groups across sites. Species classified as INCREASERS and DECREASERS follow Reid (1999), and species classified as OTHER and WOODLAND follow Seddon *et al*. (2001) (see Methods).

Influence of Habitat Characteristics on Bird Species Richness and Density/Abundance

Habtitat complexity was not related to log patch area (r2 = 0.06, F1,21 = 1.26, p = 0.274) or area-to-perimeter ratio (r2 = 0.02, F1,21 = 0.524, p = 0.477).

Patch Area

DECREASER species richness and WOODLAND species richness were both significantly positively related to log patch area (Table 18; Fig. 35, Fig. 36). INCREASER abundance was also significantly negatively related to log patch area (Table 18; Fig. 37). Overall species richness, INCREASER species richness, overall abundance, DECREASER abundance, and WOODLAND abundance were not related to log patch area (Table 18). However, significance between WOODLAND abundance and log patch area approached significance (Table 18).

Table 18. Regression statistics from tests of species richness and abundance against log patch area (ha) for different bird groups. Bird groups follow Reid (1999) and Sneddon *et al*. (2001). * denotes significance at $p < 0.05$; ** denotes significance at $p < 0.01$.

Variables	**r^2**	**$F_{1,21}$**	**p**
overall species richness vs. log patch area	0.10	2.55	0.125
DECREASER species richness vs. log patch area	0.30	8.98	0.007**
INCREASER species richness vs. log patch area	0.01	0.31	0.582
WOODLAND species richness vs. log patch area	0.29	8.63	0.008**
overall abundance vs. log patch area	0.07	1.66	0.211
DECREASER abundance vs. log patch area	0.05	1.22	0.282
INCREASER abundance vs. log patch area	0.23	6.20	0.021*
WOODLAND abundance vs. log patch area	0.15	3.64	0.070

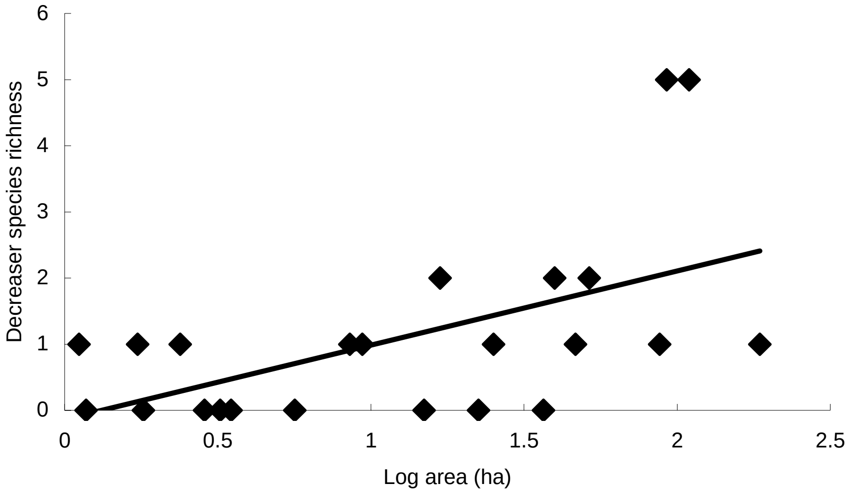


Figure 35. DECREASER species richness increases with increasing log patch area (ha). The relationship is represented by y = 1.1219x – 0.1357.

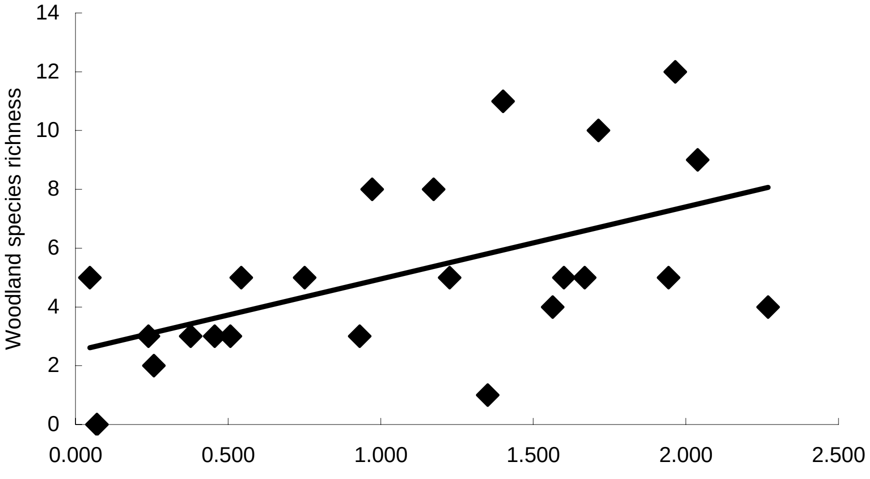


Figure 36. WOODLAND species richness increases with increasing log patch area (ha). The relationship is represented by y = 2.4524x + 2.5012.

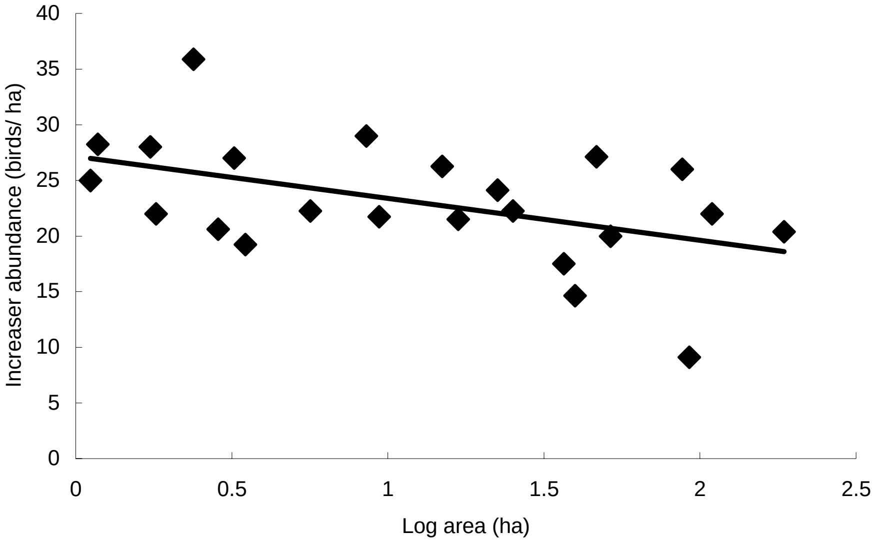


Figure 37. INCREASER abundance (birds/ ha) decreases with increasing log patch area (ha). The relationship is represented by $y = -3.7615x + 27.138$.

WOODLAND and DECREASING bird species richness increased with increasing patch size (Fig. 38). This trend was linear for WOODLAND species, and curvilinear for DECREASER species (Fig. 38).

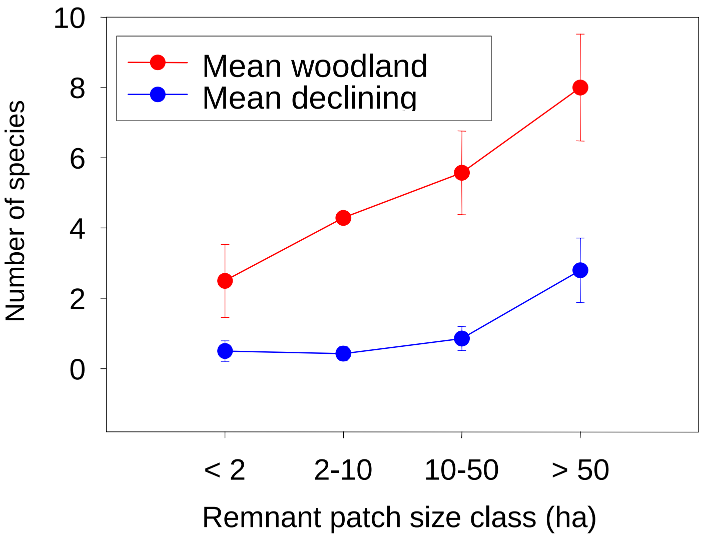


Figure 38. Bird species richness increases with increasing patch size. Categories WOODLAND and DECREASERS follow Reid (1999).

Edge Effects

DECREASER species richness and WOODLAND species richness were both significantly positively related to area-to-perimeter ratio (Table 19; Fig. 39; Fig. 40). DECREASER abundance was significantly positively related to area-to-perimeter ratio (Table 19; Fig. 41). INCREASER abundance was significantly negatively related to area-to-perimeter ratio (Table 19; Fig. 42). Overall species richness, INCREASER species richness, overall abundance, and WOODLAND abundance were not significantly related to area-to-perimeter ratio (Table 19).

Table 19. Regression statistics from tests of species richness and abundance against area-to-perimeter ratio for different bird groups. Bird groups follow Reid (1999) and Sneddon *et al*. (2001). * = significant result.

Variables	**r^2**	**$F_{1,21}$**	**p**
overall species richness vs. area: perimeter	0.07	1.60	0.219
DECREASER species richness vs. area: perimeter	0.56	26.21	0.000***
INCREASER species richness vs. area: perimeter	0.09	2.19	0.153
WOODLAND species richness vs. area: perimeter	0.23	6.17	0.022*
overall abundance vs. area: perimeter	0.08	1.82	0.192
DECREASER abundance vs. area: perimeter	0.22	5.94	0.024*
INCREASER abundance vs. area: perimeter	0.31	9.64	0.005**
WOODLAND abundance vs. area: perimeter	0.12	2.98	0.099

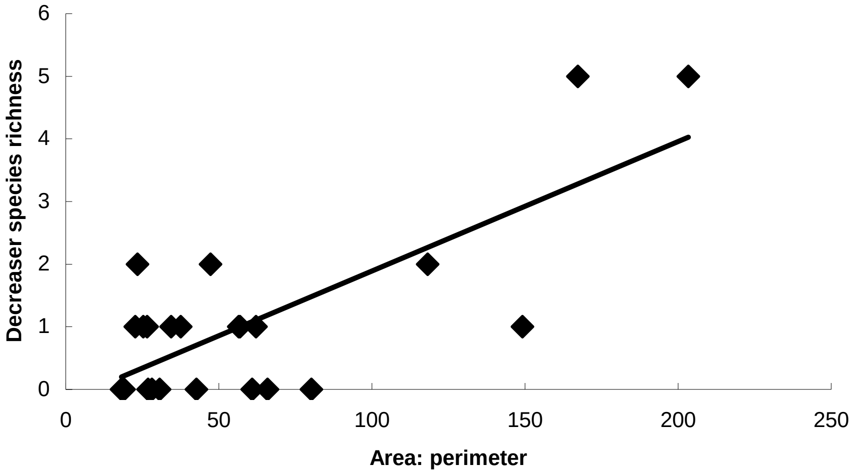


Figure 39. DECREASER species richness increases with increasing area to perimeter ratio. The relationship is represented by y = 0.0207x – 0.1746.

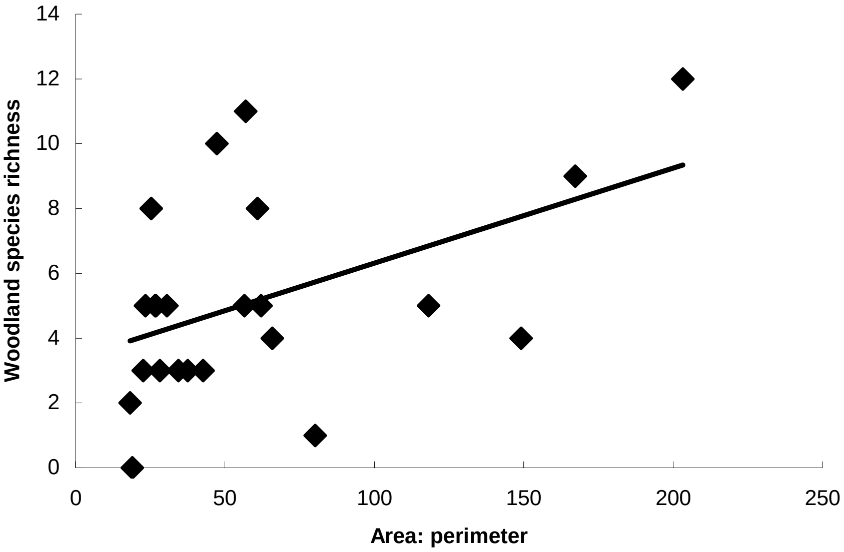


Figure 40. WOODLAND species richness increases with increasing area to perimeter ratio. The relationship is represented by y = 0.0293x + 3.3852.

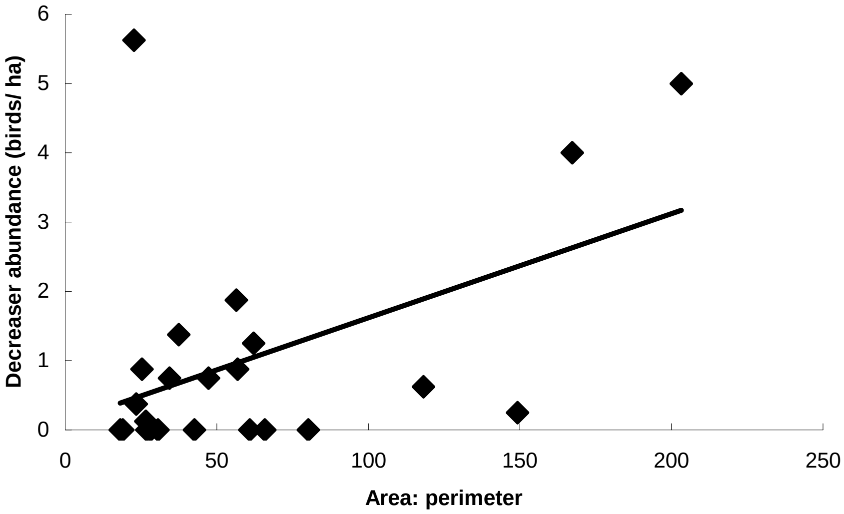


Figure 41. DECREASER abundance (birds/ ha) increases with increasing area to perimeter ratio. The relationship is represented by y = 0.01250x + 0.1150.

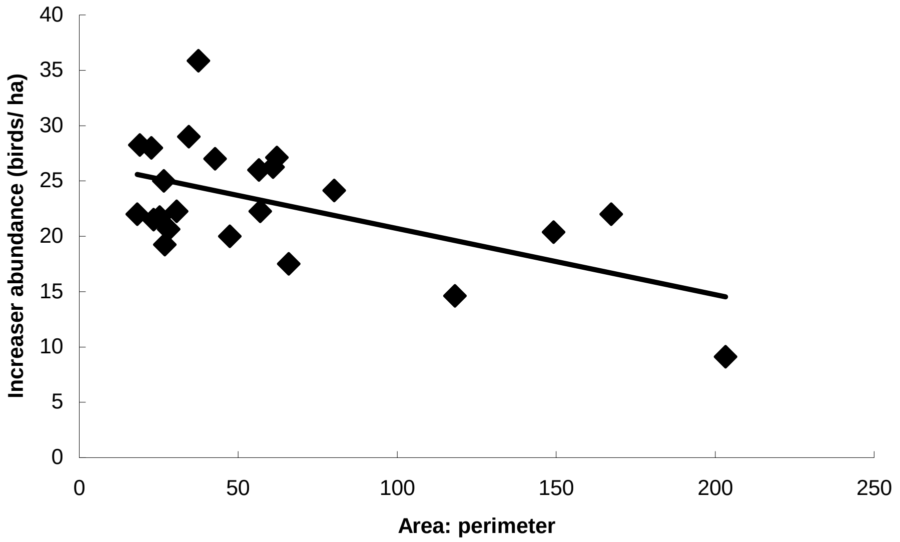


Figure 42. INCREASER abundance (birds/ ha) decreases with increasing area to perimeter ratio. The relationship is represented by y = -0.0598x + 26.688.

Habitat Complexity

WOODLAND species richness was significantly positively related to habitat complexity (Table 20; Fig. 43). WOODLAND abundance was also significantly positively related to habitat complexity (Table 20; Fig. 44). Overall species richness, DECREASER species richness, INCREASER species richness, overall abundance, DECREASER abundance, and INCREASER abundance were not related to area-to-perimeter ratio (Table 20).

Table 20. Regression statistics from tests of species richness and abundance, patch area, and area-to-perimeter ratio, against habitat complexity for different bird groups. Bird groups follow Reid (1999) and Sneddon *et al*. (2001). * = significant result.

Variables	**r^2**	**$F_{1,21}$**	**p**
OVERALL species richness vs. habitat complexity	0.10	2.28	0.146
DECREASER species richness vs. habitat complexity	0.12	2.73	0.113
INCREASER species richness vs. habitat complexity	0.01	0.24	0.628
WOODLAND species richness vs. habitat complexity	0.19	4.88	0.038*
OVERALL abundance vs. habitat complexity	0.02	0.48	0.497
DECREASER abundance vs. habitat complexity	0.03	0.63	0.438
INCREASER abundance vs. habitat complexity	0.02	0.50	0.485
WOODLAND abundance vs. habitat complexity	0.19	4.93	0.038*
log area vs. habitat complexity	0.06	1.26	0.274
area: perimeter vs. habitat complexity	0.02	0.52	0.477

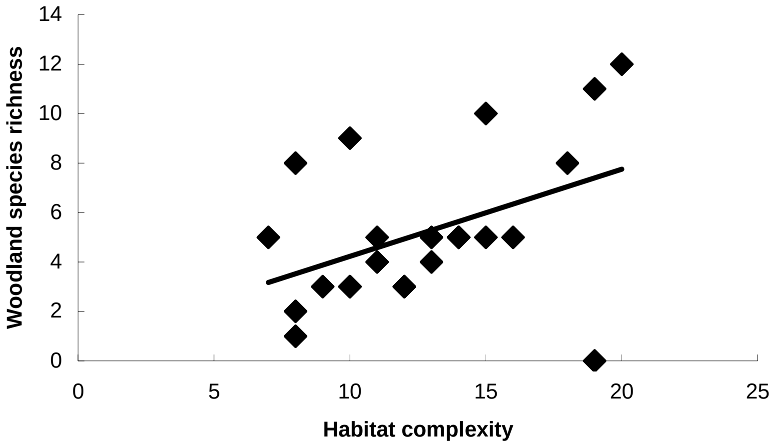


Figure 43. WOODLAND bird species richness increases with increasing habitat complexity. The relationship is represented by y=0.3525x + 0.6989.

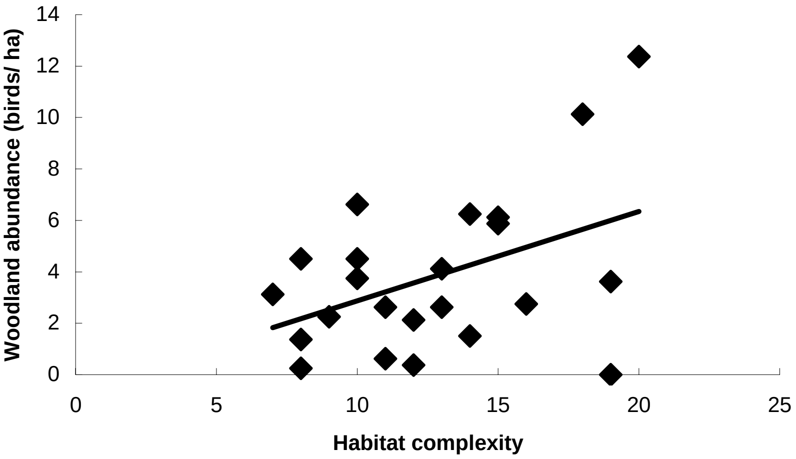


Figure 44. WOODLAND bird species abundance (birds/ ha) increases with increasing habitat complexity. The relationship is represented by y=0.3476x - 0.6091.

Habitat Variables and Species Diversity

Bird diversity (Shannon-Wiener diversity index) and log patch area were positively related, but the relationship only approached significance (F1,21 = 3.96; p = 0.060). Bird diversity was not significantly related to area-to-perimeter ratio (F1,21 = 1.84; p = 0.189), or to habitat complexity (F1,21 = 1.0754; p = 0.312).

Influence of Landscape and Habitat Characteristics on Bird Species Richness and Density/Abundance

A multiple regression model was significant in predicting overall species richness (r2 = 0.40; F1,21 = 2.97; p = 0.048). The number of patches >10ha within a 2km radius, canopy cover of vegetation 2-4m, number of dead trees and the number of regenerating trees (<10cm DABH) were all significant predictors in the model (Table 21). Canopy cover of vegetation 2-4m was positively related to overall species richness, whilst the remaining three variables were inversely related to overall species richness (Table 21).

No significant model was developed relating landscape and habitat variables to overall bird species abundance.

Table 21. Multiple regression model results examining the influence of habitat and landscape variables on bird species richness. Only significant or nearly significant variables are included.

Variable	**Parameter estimate**	**SE**	**Type II SS**	**F**	**p**
Intercept	26.1400	2.4482	2174.2349	114.04	<0.0001
2 km NOP	-0.7374	0.3552	82.1689	4.31	0.0525
CCS (0.5-2m)	0.2326	0.0784	167.6426	8.79	0.0083
Dead T	-1.1894	0.4849	114.7131	6.02	0.0246
Regen	-0.0685	0.0331	81.8753	4.29	0.0529

A step-wise multiple regression including patch area indicated that none of the four measures of isolation (distance to nearest patch > 10 ha, distance to nearest patch > 100 ha, distance to nearest patch of same vegetation type > 10 ha, number of patches > 10 ha in a 2 km radius) were found to have a significant effect on OVERALL, WOODLAND, DECREASER or INCREASER species richness (all p > 0.05). The models indicated that patch area was significantly inversely related to species richness of DECREASERS (r2= 0.32, F1,21 = 10.06; p= 0.005) and WOODLAND birds (F 1,21= 7.84; r2= 0.272; p= 0.011). Patch area was not significantly related to INCREASERS species richness or OVERALL bird richness (all p > 0.05).

The Influence of Noisy Miner Density on Birds

Noisy minor (Manorina melanocephala) abundance was not related to richness or abundance of OVERALL species, DECREASERS, INCREASERS, or WOODLAND species (Table 22). Noisy minor (Manorina melanocephala) abundance was also not related to log patch area or area-to-perimeter ratio (Table 22).

Table 22. Regression statistics from tests of the relationships between noisy miner (*Manorina melanocephala*) abundance (birds/ha) and species richness, abundance, patch area, and area-to-perimeter ratio. Bird groups follow Reid (1999) and Sneddon *et al*. (2001).

Variables	r^2	$F_{1,21}$	**p**
log area versus noisy miner abundance	0.13	3.01	0.097
area: perimeter versus noisy miner abundance	0.00	0.02	0.939
DECREASER species richness vs. noisy miner abundance	0.06	1.31	0.266
INCREASER species richness vs. noisy miner abundance	0.01	0.24	0.630
WOODLAND species richness vs. noisy miner abundance	0.00	0.09	0.772
DECREASER abundance vs. noisy miner abundance	0.15	3.64	0.070
INCREASER abundance vs. noisy miner abundance	0.00	0.00	0.963
WOODLAND abundance vs. noisy miner abundance	0.00	0.08	0.785

B. Habitat Utilisation, Movements, and Diet of Long-necked Turtles on a Rice Farm

Radio-telemetry Data

Radio-tagged turtles (7 males, 7 females) were located 503 times between September 2001 and late March 2002 (males = 236 fixes, females = 267 fixes). These data formed the basis of an assessment of habitat use and movement patterns. Laparoscopy conducted on the radio-tagged turtles at the completion of the study revealed that all turtles were sexually mature and in reproductive condition with the exception of one male (No. 1). The sexual maturity status of this male was uncertain. Of the fourteen individuals radio-tracked, the movements and habitat use of twelve were successfully followed from September to March (Fig.'s 45 - 56). Two males, No. 8 (originally caught in the supply channel) and No. 4 (originally caught in the farm channel), died in January and February, respectively. The remains of both were found in the supply channel and there was no evidence to suggest the cause of death. The movements and habitat use recorded for these turtles in the two months prior to their death were discarded, as the possibility of sickness prior to death may have yielded unrepresentative data. Radio-transmitter failure was experienced for one male (No. 10) during November and December. Consequently, no data could be collected for this individual during these months.

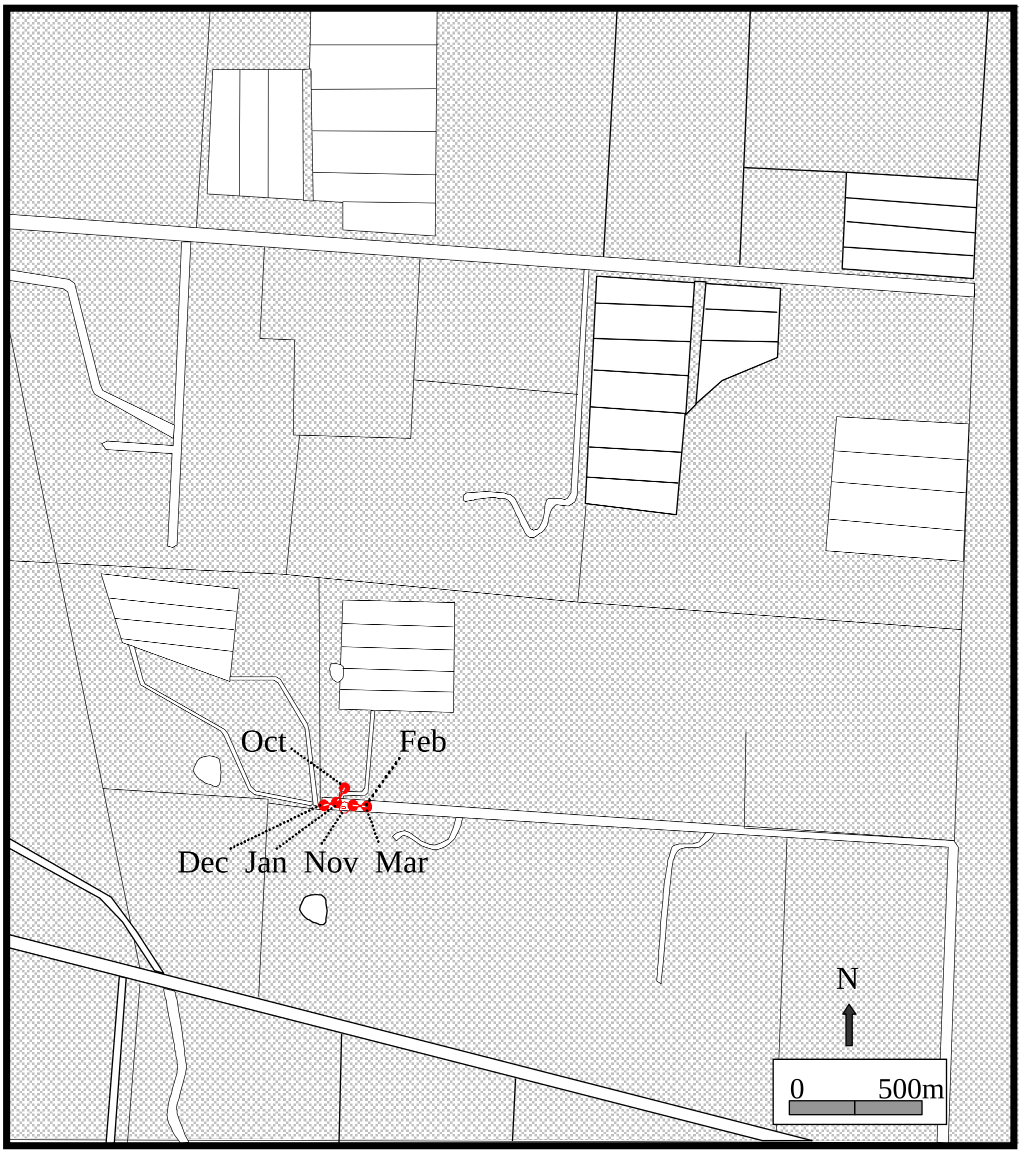


Figure 45. Radio-telemetry locations (n = 55) from a male *C. longicollis* (No. 1) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

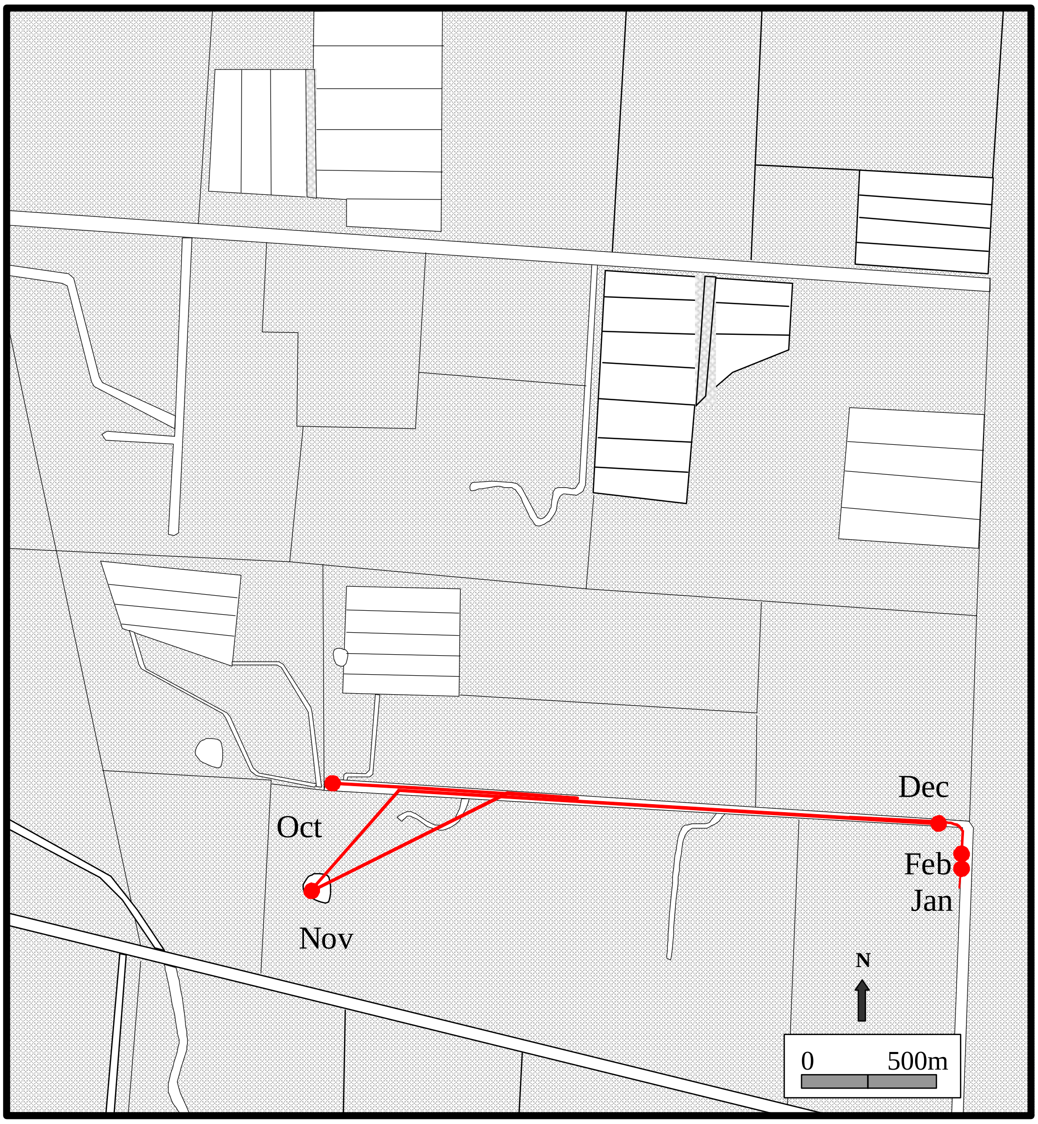


Figure 46. Radio-telemetry relocations (n = 50) for a male *C. longicollis* (No. 2) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

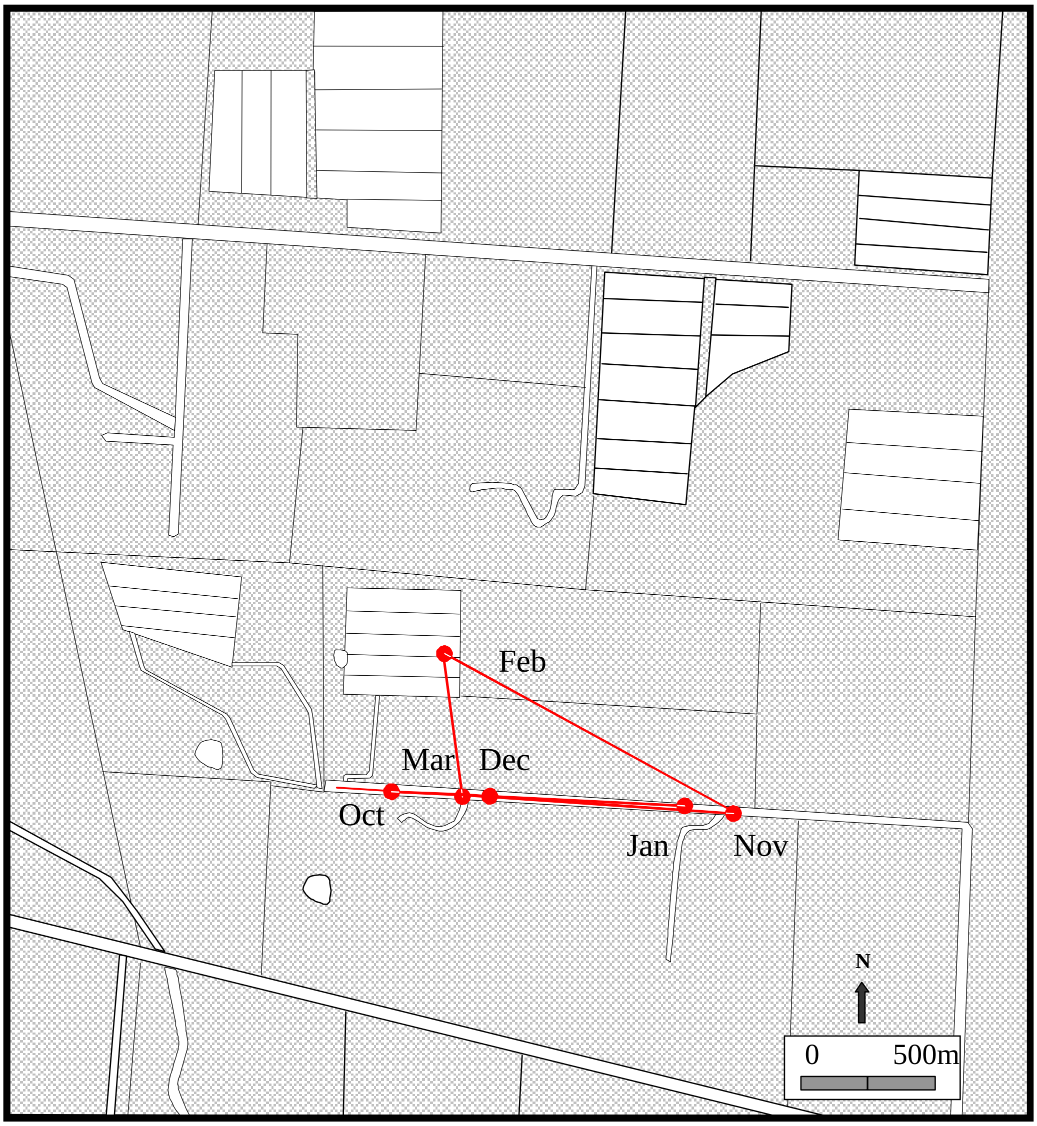


Figure 47. Radio-telemetry relocations (n = 48) for a female *C. longicollis* (No. 3) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

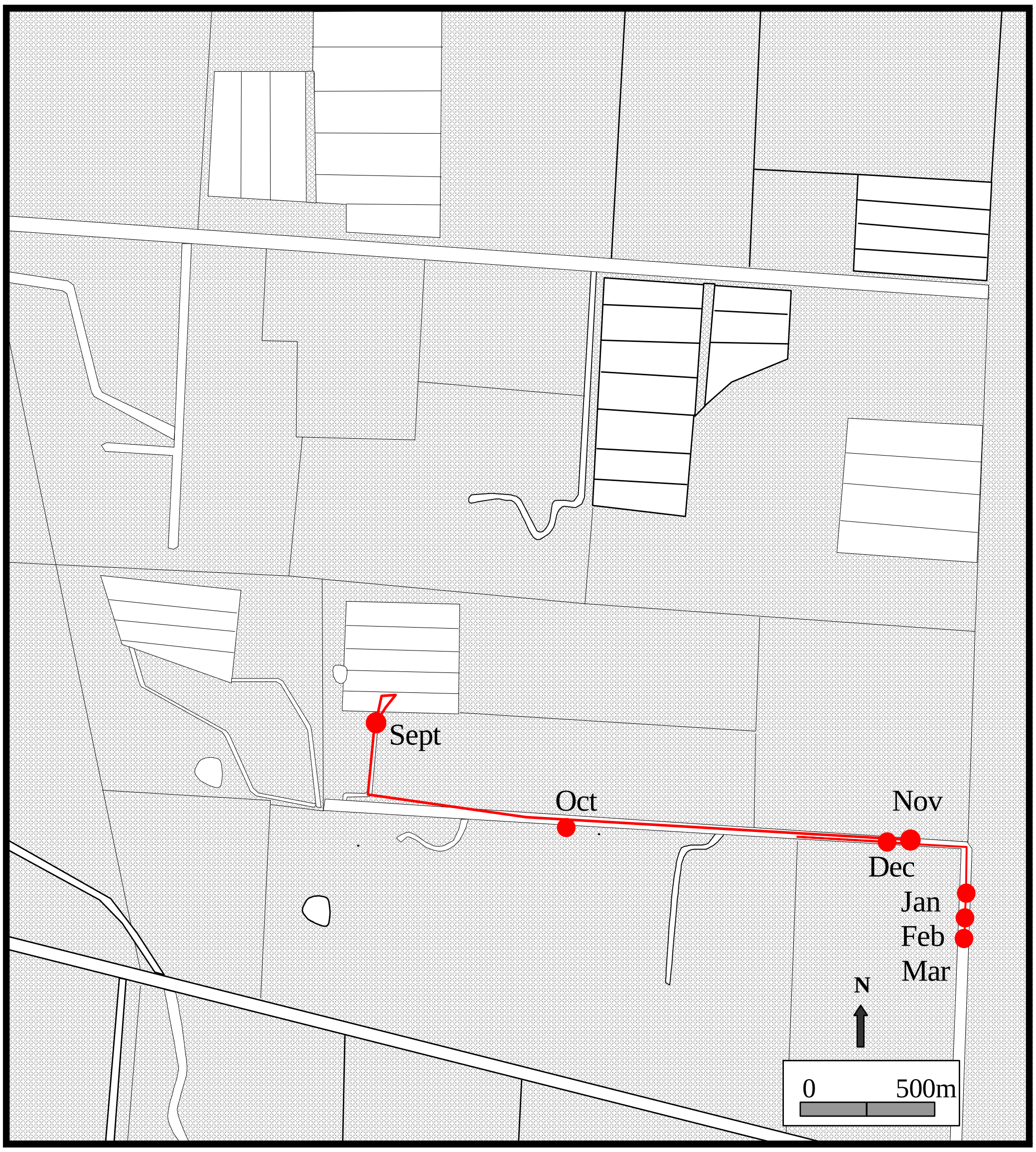


Figure 48. Radio-telemetry relocations (n = 48) for a male *C. longicollis* (No. 5) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

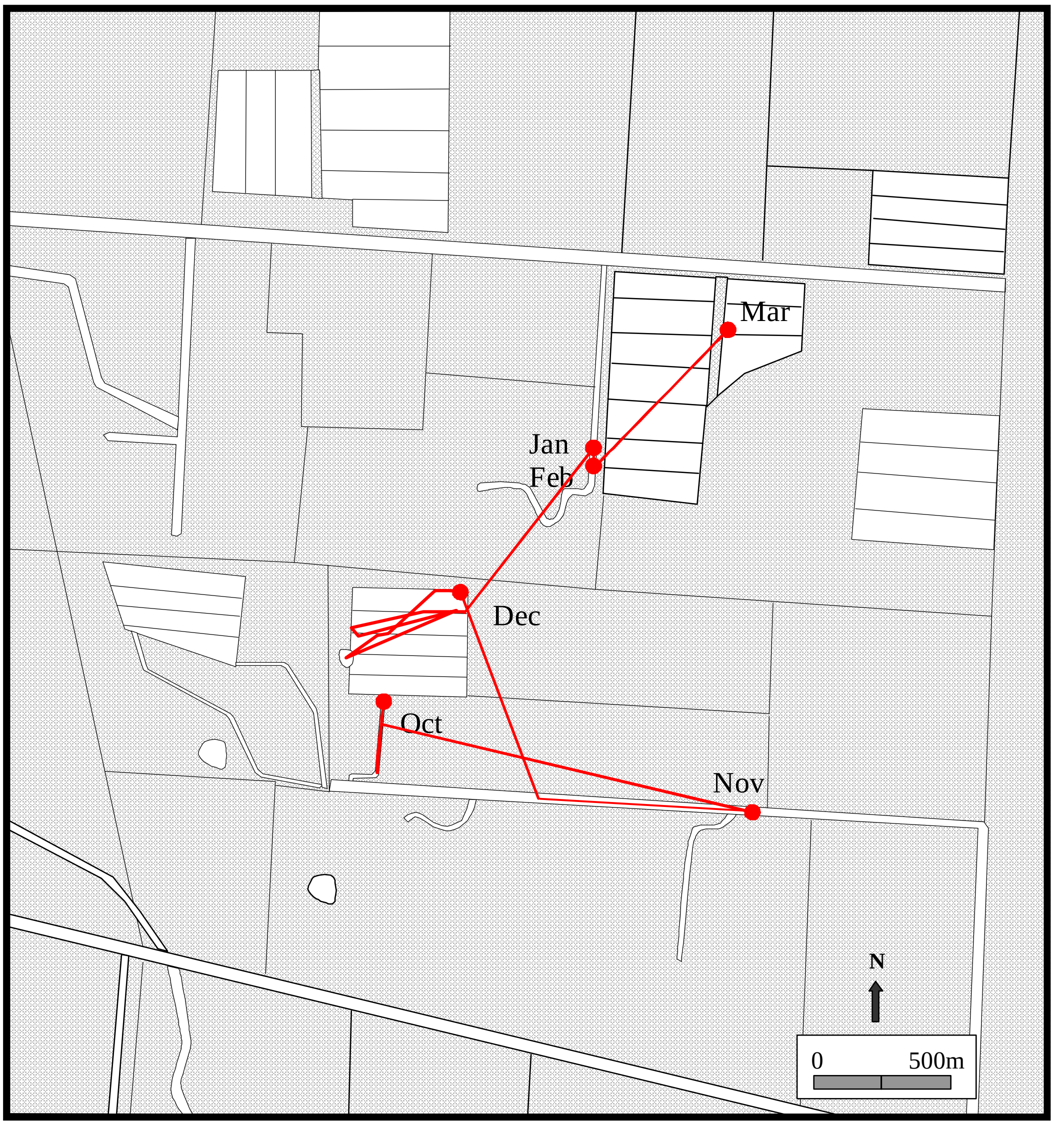


Figure 49. Radio-telemetry relocations (n = 44) of a female C. longicollis (No. 6) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

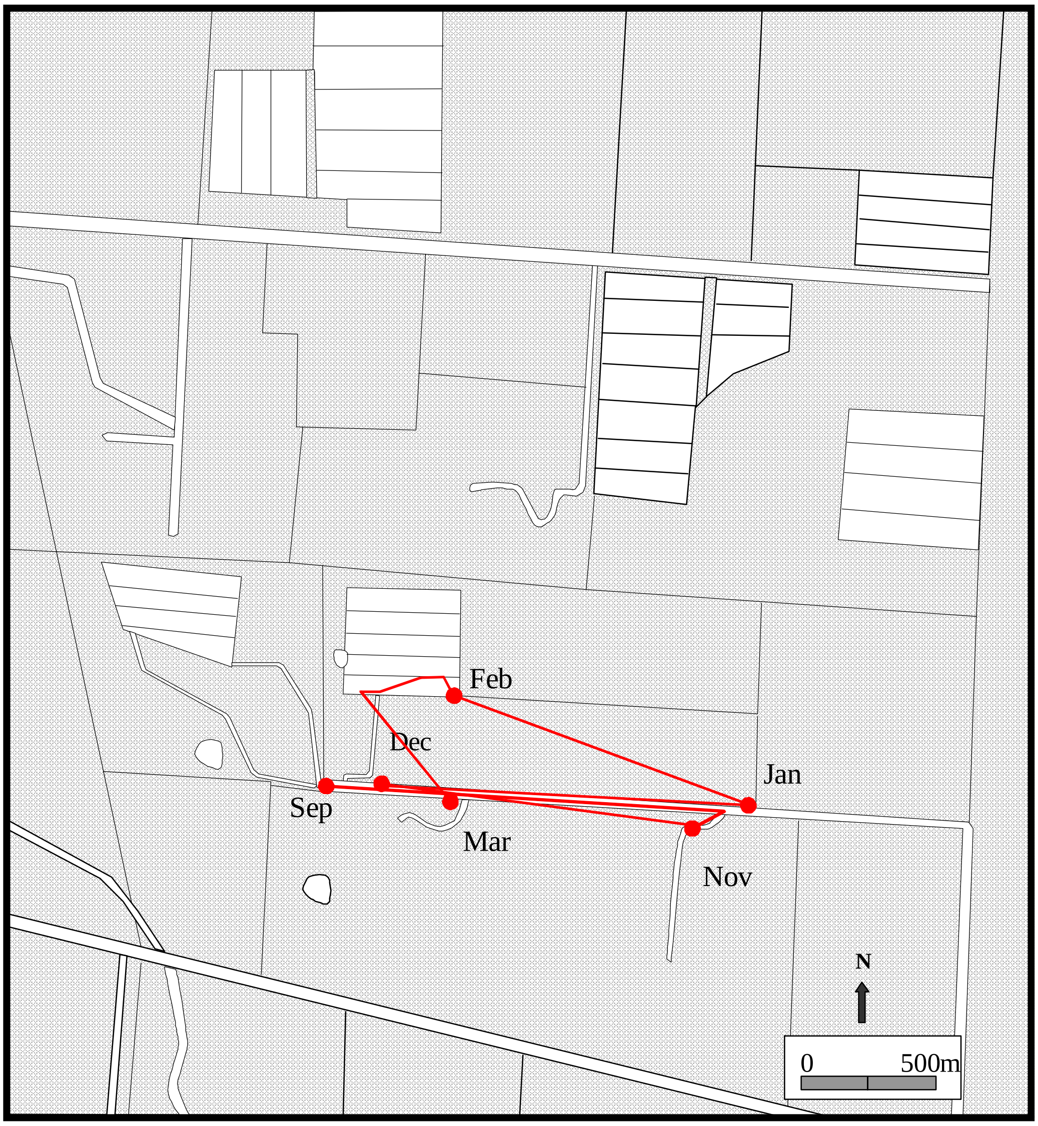


Figure 50. Radio-telemetry relocations (n = 41) of a female *C. longicollis* (No. 7) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

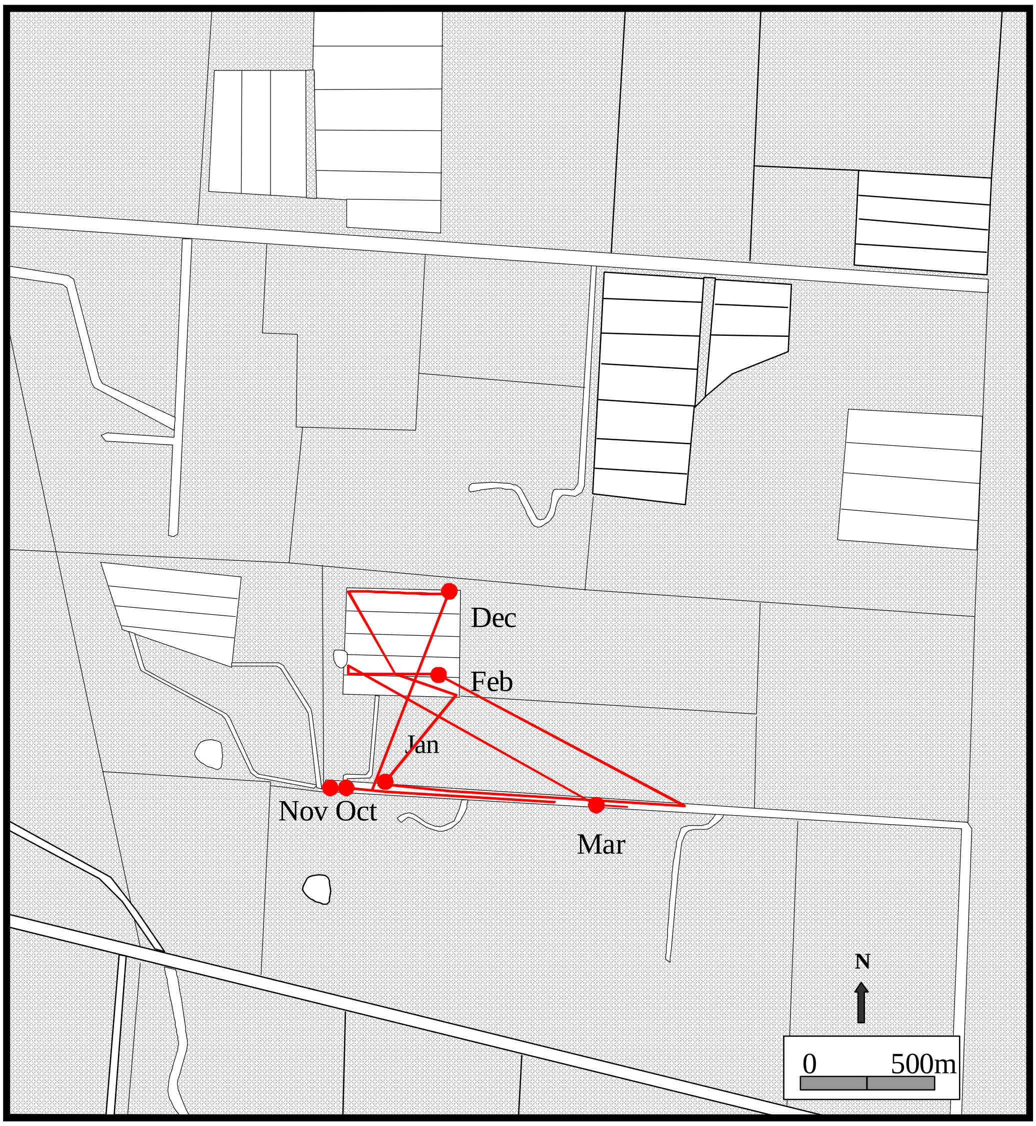


Figure 51. Radio-telemetry relocations (n = 35) of a female *C. longicollis* (No. 9) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

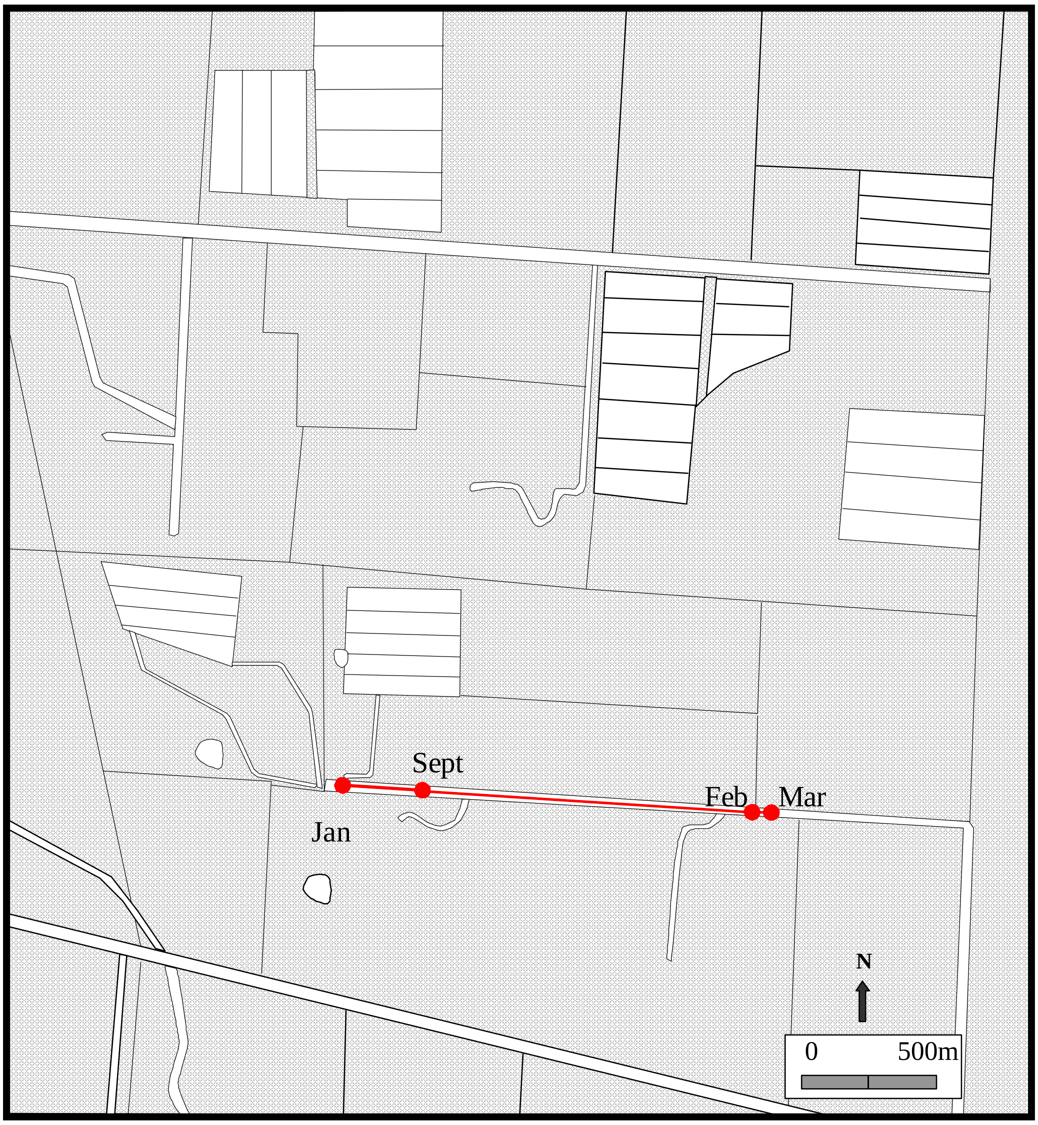


Figure 52. Radio-telemetry relocations (n = 20) of a male *C. longicollis* (No. 10) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

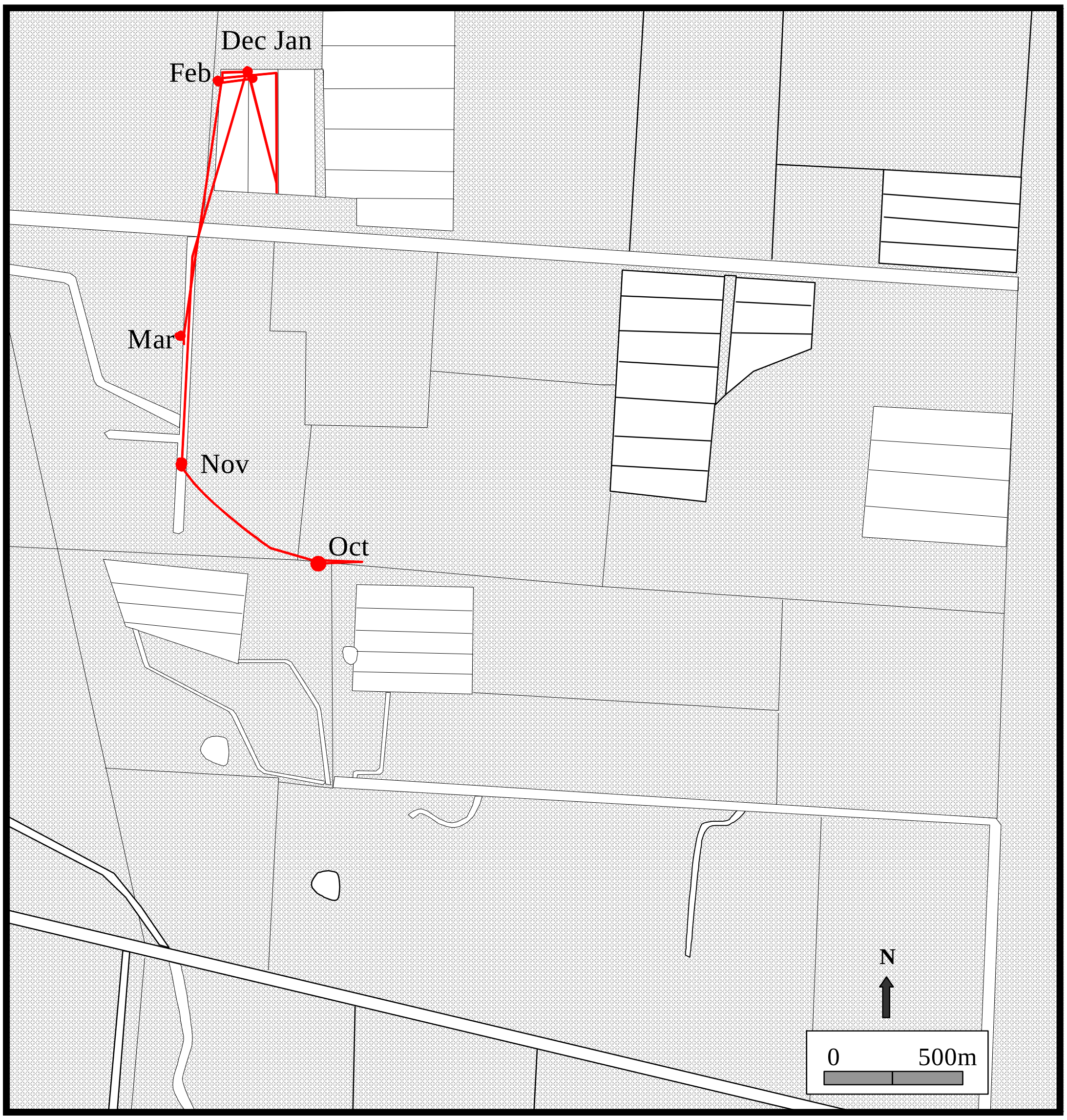


Figure 53. Radio-telemetry relocations (n = 35) of a male *C. longicollis* (No. 11) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

Figure 54. Radio-telemetry relocations (n = 34) of a female *C. longicollis* (No. 12) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

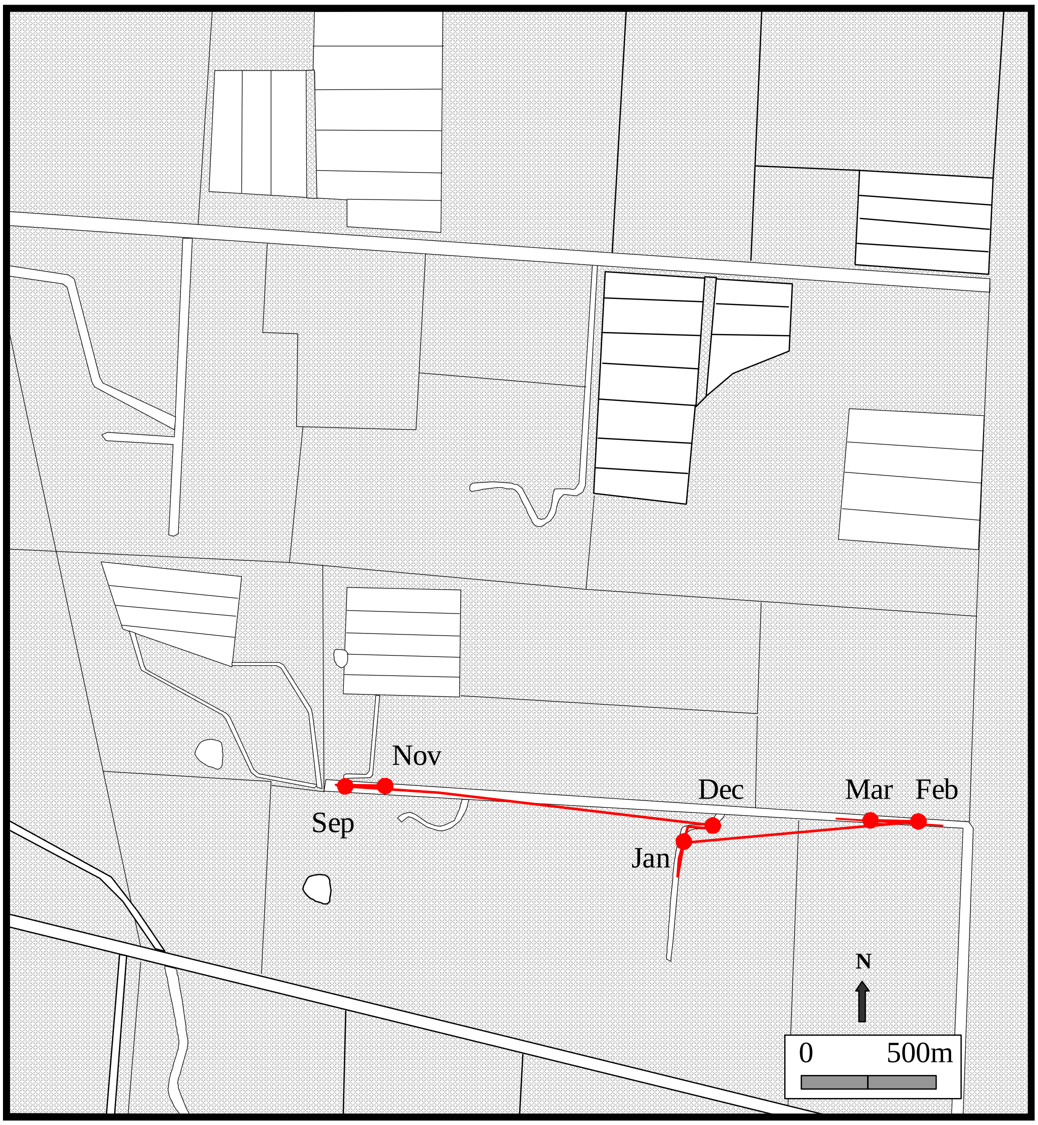


Figure 55. Radio-telemetry relocations (n = 30) of a female *C. longicollis* (No. 13) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

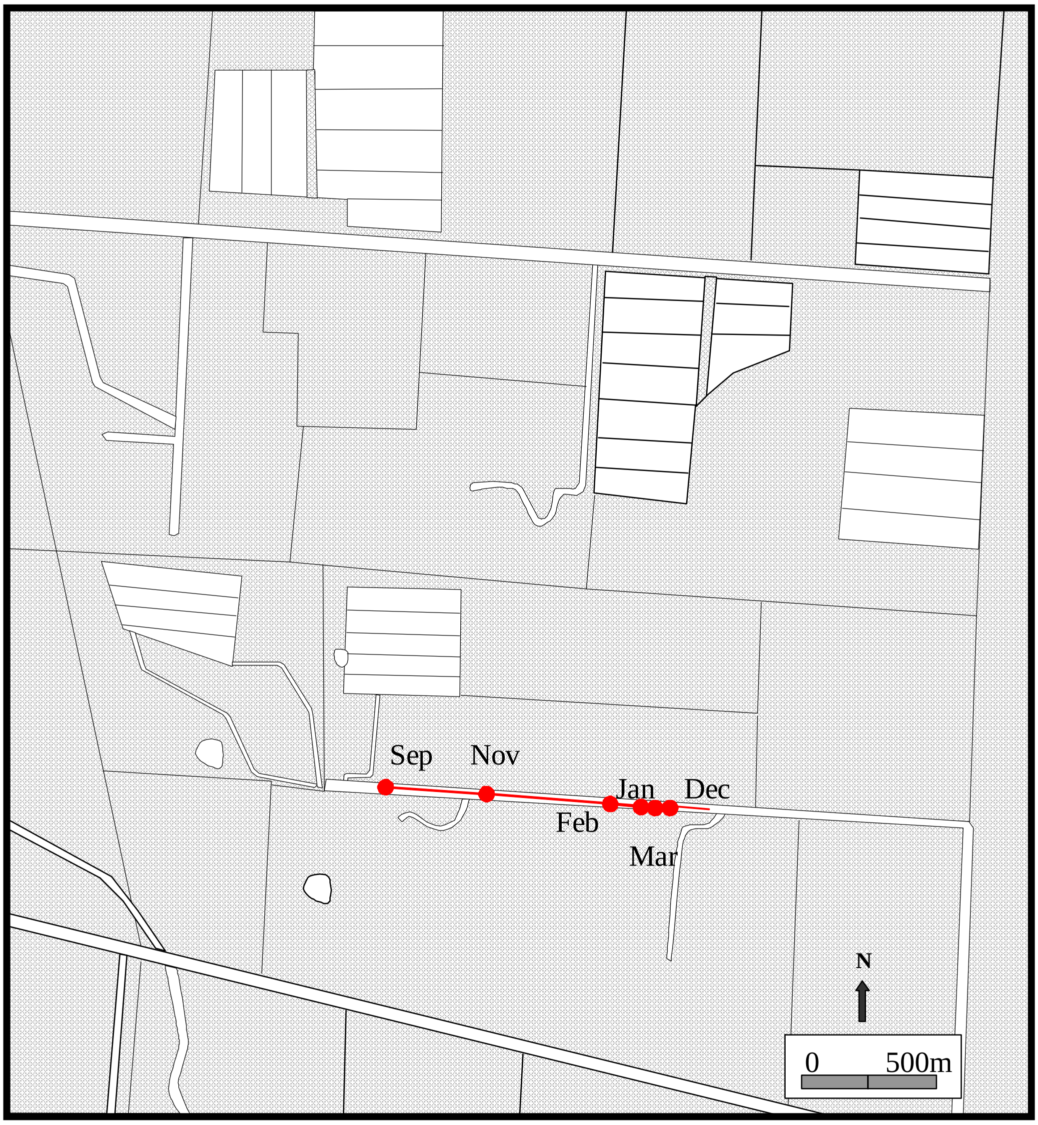


Figure 56. Radio-telemetry relocations (n = 35) of a female *C. longicollis* (No. 14) from September 2001 to March 2002. Points represent the first recorded location during each sampling period, with lines connecting points in chronological order.

Population Size

Ninety-seven turtles were captured and marked within the Marsden property, of which 43 were captured in the rice paddies (7 by trapping and 36 by muddling), 39 in the supply channel (35 by trapping and 4 by hand), eight in the farm channel (7 by traps and 1 by hand), six in the dam (all by trapping), and one was captured by hand on a roadside. Eight of the 97 marked turtles were recaptured once and one was recaptured twice. This resulted in a recapture rate of 9.1%. Population size estimation using the Jolly-Seber method for open populations was unsuccessful due to the low recapture rate as well as the short periods between the initial capture and recapture of the turtles (Seber, 1973; Caughley, 1980). The minimum population size estimate for the 1.28 km^2 property is 97 turtles (based on the number of captures).

Sex Classification by Vertical Displacement

The vertical displacement to plastron length ratio (VD/PL) frequency distribution plotted for turtles larger than 10 cm ($n = 67$) had a bimodal distribution (Fig. 57). The data indicate that females generally had VD/PL ratios of greater than 0.091 and males less than 0.091. Classification of sex based on the VD/PL ratio cut-off in the bimodal distribution (males < 0.091 and females > 0.091) yielded an accuracy of 91% for the twelve radio-tagged turtles that were laparoscoped. Laparoscopy revealed that there is some overlap within the VD/PL sex classification method. One male had a VD/PL ratio of 0.108.

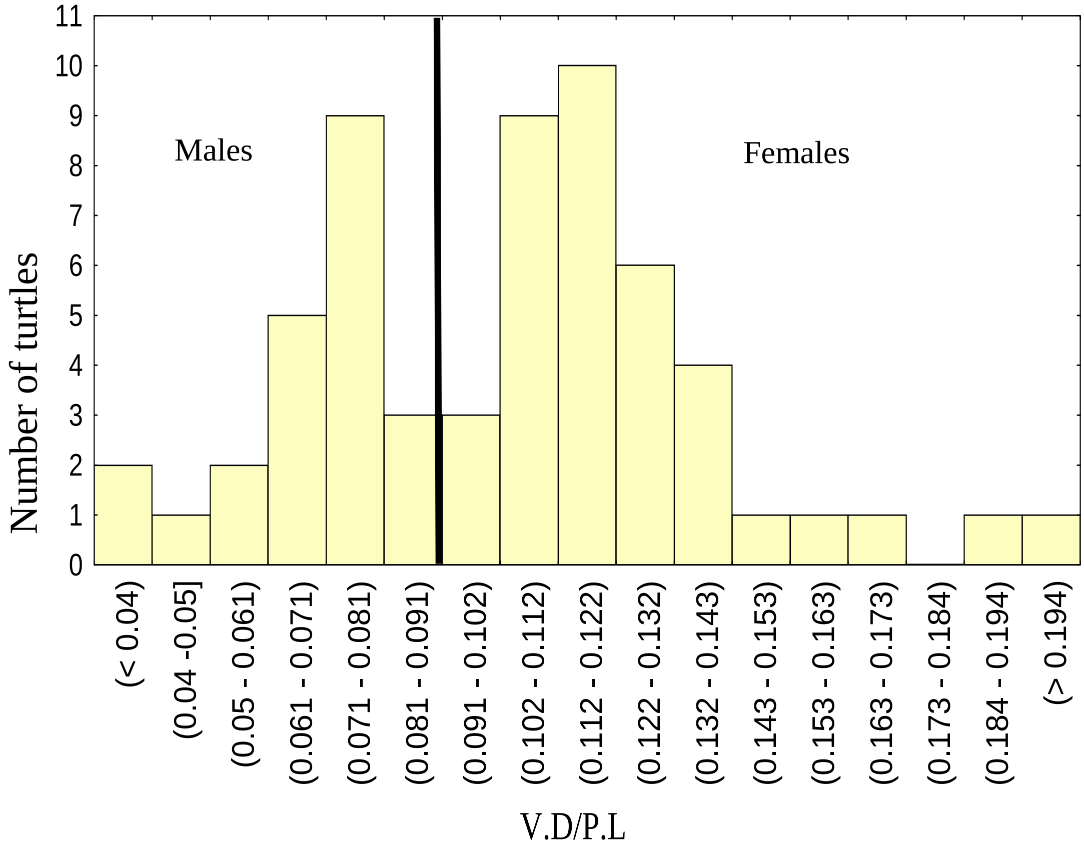


Figure 57. Frequency distribution of the ratio of vertical displacement to plastron length (VD/PL) for *C. longicollis* larger than 10 cm carapace length. The vertical displacement (VD) is defined as the vertical distance between the carapace and the plastron at the anal notch of the turtle. This value was divided by the plastron length (PL) to give a standardised ratio of vertical displacement. The bimodal distribution bisected by the thick black line, reflects a difference between sexes, with females having greater VD/PL ratios.

Habitat Utilisation

The supply channel was the habitat utilised most extensively by the telemetered turtles, containing 347 (69%) of the total daily relocations. Habitat utilisation in the farm channels and rice bays was less extensive, containing 80 (16 %) and 60 (12 %) of the total relocations, respectively. The farm dam was the habitat least used, with 15 (3 %) of total daily relocations. Figure 58 shows habitat utilisation by telemetered turtles throughout the study, broken down by month.

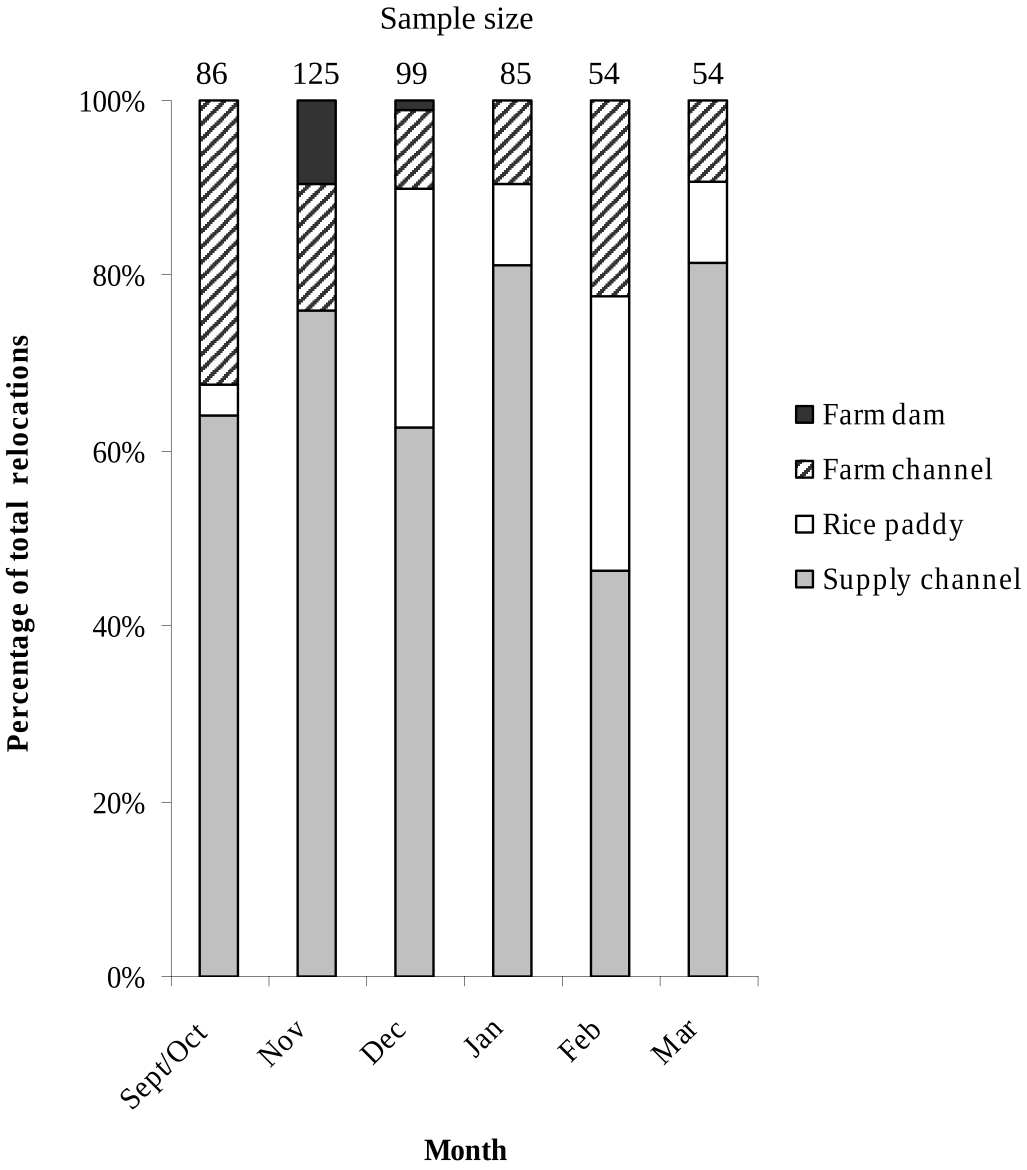


Figure 58. Habitat use by radio-tagged *C. longicollis* (7 females and 5 males) from September 2001 to late March 2002, presented as the percentage of total radio-telemetry fixes collected in each month that were in each major habitat type. Numbers above columns are sample sizes.

Supply and Farm Channels

All telemetered turtles (except one male = No. 11) occupied the supply channel at some time between September and late March, and non-telemetered turtles were captured from the supply channel in all months sampled: September/October (14 turtles), November, (6 turtles), December (3 turtles), January (7 turtles), February (3 turtles) and March (6 turtles). Two radio-tagged turtles used only supply channels during the study period (No. 10 and No. 14; Fig. 52, 55), and five tagged turtles used only supply or farm channels (No. 1, No. 10, No. 12, No. 13, No. 14; Fig. 45, 51, 53, 54, 55). Six of the twelve radio-tagged turtles occupied the smaller farm channels at some point during September to late March. The greatest use of farm channels by telemetered turtles occurred in September/October (n = 86 fixes = when 33% of the total daily fixes; Fig. 58). No radio-tagged turtles used only farm channels.

Rice Bays

Flooding of the rice bays commenced on 29 September 2001 and took approximately five days to complete. One radio-tagged turtle moved into the rice bays during the flooding process, and four non-telemetered turtles were captured at the irrigation water inlet of the rice bays during flooding. Six of the 12 turtles radio-tagged turtles used the rice bay habitat at some point between September and March. Four telemetered turtles occupied rice habitats in February, and three occupied rice in December (Fig. 59). Rice bays were used by at least some telemetered turtles in all months except November (Fig. 59). However, 21 non-telemetered turtles were hand-captured in the rice tofos in November. Two females (No. 6 and No. 9) moved between rice bays and irrigation channels twice within the study period.

Draining of rice bays began on 14 March 2002. Prior to this, four telemetered turtles occupied the bays. These turtles all returned to the irrigation channel from which they originated before the bay water depth was reduced to approximately 15 cm on 17 March. On 17 March, 2002, the complete length of each tofo on the farm was searched thoroughly for turtles by muddling, but no turtles were captured. One female (No. 6) occupied a set of rice bays on a farm northeast of the study site during March (Fig. 49). However, these bays were not drained until April.

Within rice bays, all relocations of telemetered turtles were in the rice tofos, and never in the rice field (crop). However, eight individuals captured in the rice tofos and followed by spool-tracking in January used both the rice tofo and rice field. Typically, spool-tracked turtles made short (2-10 m) forays into the thicker crop, invariably returning to the tofos prior to being relocated. Both adult and juvenile individuals of C. longicollis were captured in rice bays, with sizes ranging from 8.9 cm to 25.3 cm.

Farm Dams

The farm dam was the least used habitat type (Fig. 58). Only two telemetered turtles utilised farm dams. One turtle (No. 6 = female) occupied a farm dam for less than 24 hours before returning to the rice bays. Another turtle (No. 2 = male) moved from the supply channel to a dam on an adjacent dairy farm in late November and returned to the supply channel by mid-December. Early in October, six juveniles of less than 110 mm in carapace length were captured in the dam adjacent to the rice. However, no adults were captured in this dam, and no turtle of any size was captured there after October despite a sustained trapping effort. The dam dried completely in mid-January, and did not refill until February.

Movement Patterns

Radio-tagged turtles moved between habitat types in all months sampled, but displayed two distinctive peaks, one in Nov/Dec and another in Feb/Mar (Fig. 59). The second peak in movements coincided with the drainage of the rice bays. During this period the majority of total movements (n = 6) were to the supply channel (n = 4) (Fig. 59). The single movement to a rice bay in Feb/Mar (Fig. 59) was a turtle on an adjacent rice farm, where rice bays were drained after the movement (later than rice bays on the main study farm).

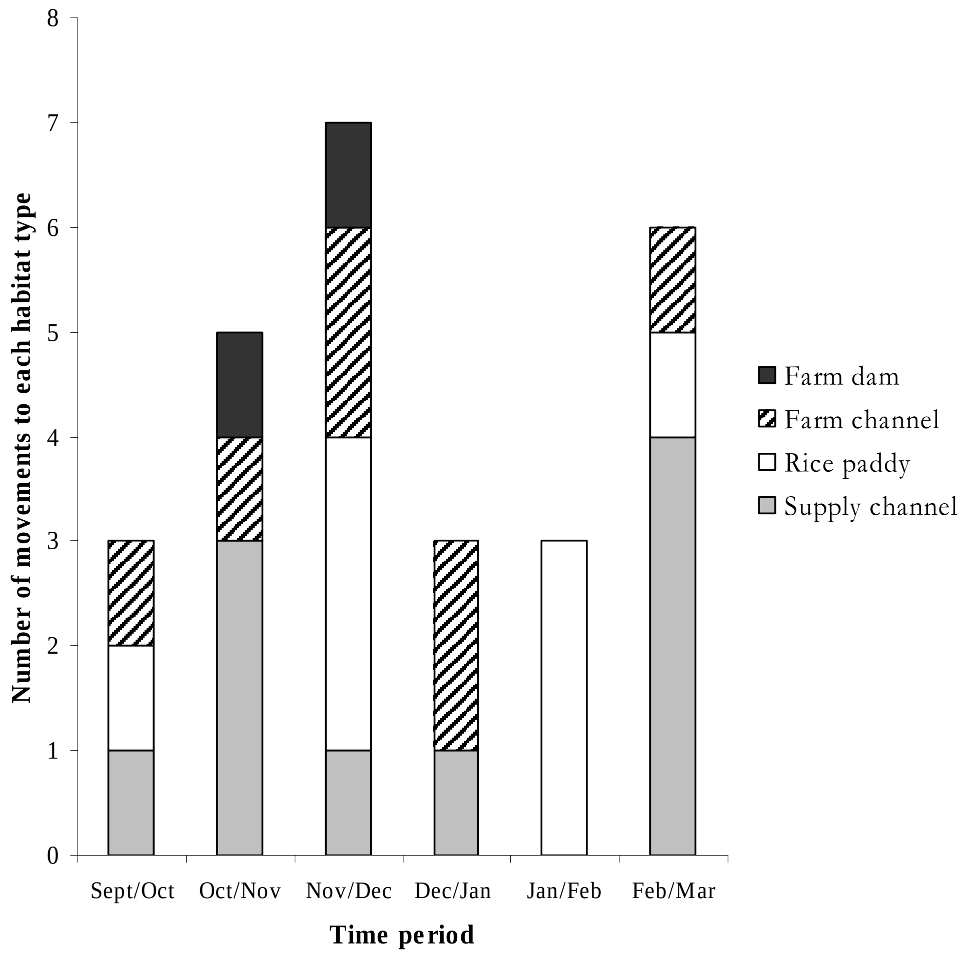


Figure 59. Number of movements made to each major habitat type by 12 radio-tagged *C. longicollis* (7 females, 5 males) in monthly periods from September 2001 to late March 2002. Each time period covers the second half of the first month and the first half of the second month. Habitats indicated represent the destination habitats.

Breaking the timing of movements down by sex, males tended to move more during Sept-Nov, while females generally moved more during Nov-Feb (Fig. 60). The female bias in movements during Feb/Mar (Fig. 60) was related to a female bias in radio-tagged turtles occupying rice habitats during that time.

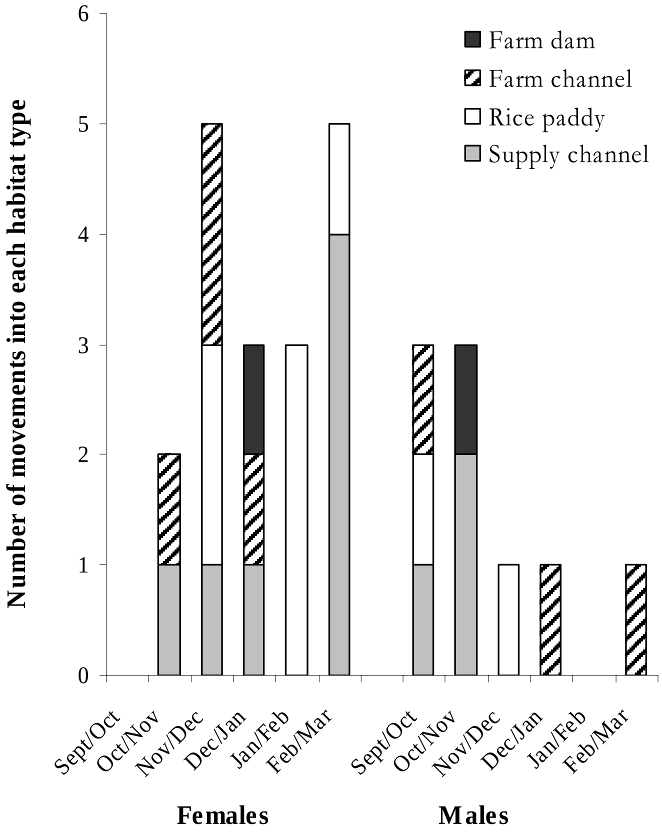


Figure 60. Number of movements made into each major habitat by radio-tagged *C. longicollis,* broken down by sex (7 females, 5 males), in monthly periods from September 2001 to late March 2002. Each time period covers the second half of the first month and the first half of the second month. Habitats indicated represent the destination habitats.

Mean daily distance between relocations was 72 ± 4.9 and ranged from < 5 m to an 887 m journey by a female in the supply channel over a 34-hour period during January (Table 23). Sex did not influence mean daily distance travelled per month (F1, 481 = 2.9; p = 0.892), or mean daily distance travelled per reproductive season (F1,481 = 0.17; p = 0.684). Reproductive seasons were as defined as: mating (September/October), nesting (November to December) and post nesting (January to March) (after Parmenter, 1976; Chessman, 1978).

Table 23. Mean daily movement distance of male (n = 5) and female (n = 7) *longicollis* from September 2001 to late March 2002 based on radio-telemetry data. Values are mean m/day ± 1 S.E. Number of movements is in parentheses.

Month	**Males**	**Females**	**Both sexes**
September/October	77 ±12.9 (56)	112 ±19.6 (24)	88 ±10.8 (80)
November	30 ±13.8 (46)	72 ±17.5 (54)	53 ±11.5 (100)
December	64 ±14.2 (40)	95 ±15.3 (56)	82 ±10.8 (96)
January	70 ±19.4 (31)	88 ±19 (47)	86 ±13.7 (78)
February	70 ±11.2 (15)	51 ±16.3 (20)	59 ±10.4 (35)
March	41 ± 7.9 (17)	40 ± 9.8 (26)	40 ± 6.6 (43)
all months	59 ± 6.3 (205)	80 ± 7.5 (227)	72 ± 4.9 (434)

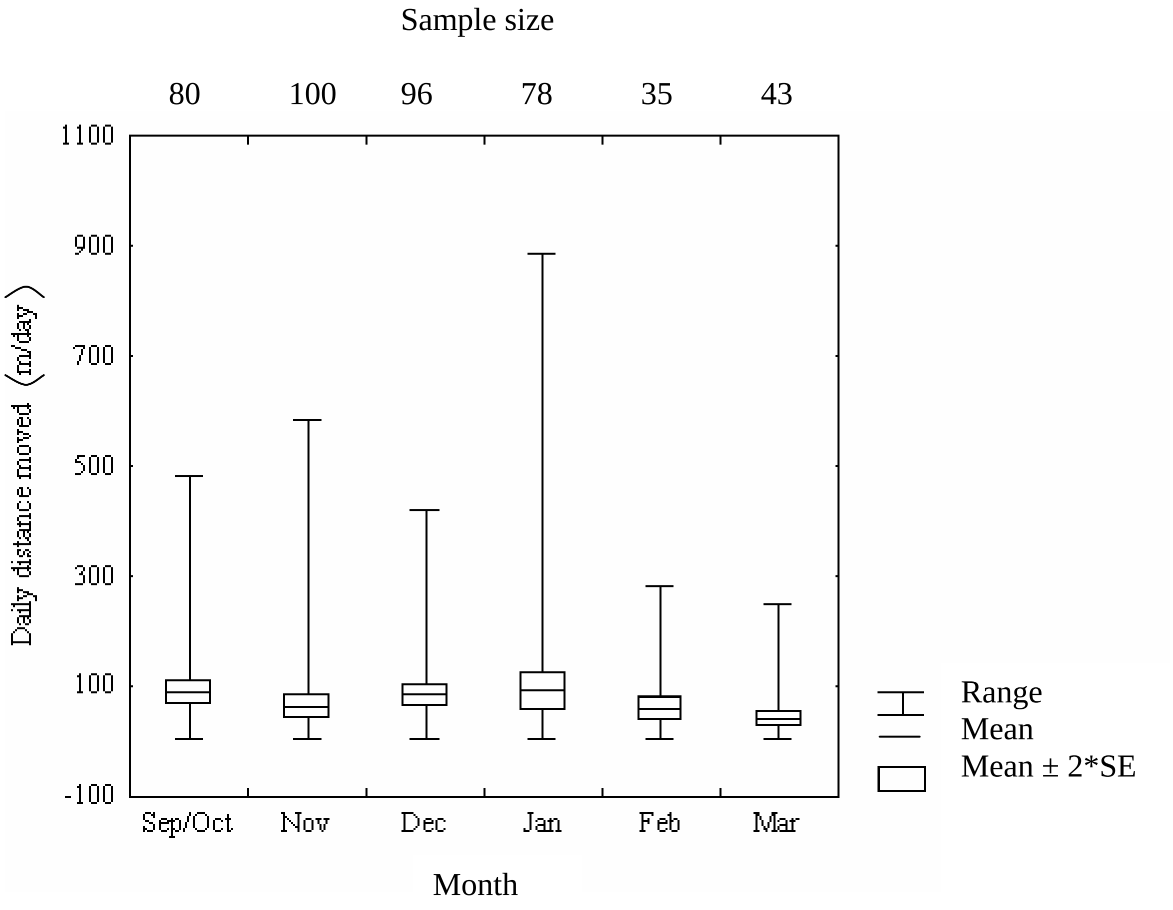


Figure 61. Pattern of average daily distance moved by C. longicollis (both sexes combined, males = 5, females = 7), plotted as a monthly average from September 2001 to late March 2002, based on radio-telemetry data.

The mean daily distance travelled was highest for both females and males in September/October when the means were 112 m/day for females and 77 m/day for males (Table 23; Fig. 61). Mean daily movement distances of both sexes combined differed significantly among months (F5,481 = 3.13, p = 0.009). A Tukey-Kramer test revealed that movement distances in September/October were significantly greater than in November (p = 0.006) and March (p = 0.031). There were no significant differences between any other months (all p > 0.05). Reproductive season also had a significant influence on movement distances of both sexes combined (F2,481 = 10.05, p< 0.0001). A Tukey-Kramer test revealed that movement distances in the mating season (September/October) were significantly greater than during the nesting season (p < 0.0001) and post-nesting period (p < 0.0001).

The high daily distances travelled in September/October coincided with a relatively high amount of rainfall in early October (73.6 mm) (Table 23). However, mean daily movement distance by month was not significantly related to mean monthly rainfall (r2 = 0.45; F1,7 = 5.4, p = 0.091).

Terrestrial Movements

The terrestrial movements of two individuals were particularly noteworthy due to the distance traversed. First, a male (No. 2) undertook a 430 m terrestrial journey from the supply channel to a dam on an adjacent dairy farm in late October, making the return journey in late November (Fig. 46). Second, a female (No.6) moved approximately 800 m from the rice bays on the Marsden farm to a farm channel on an adjacent rice farm to the northeast in late December (Fig. 49). Although the exact route she followed is unknown, the journey involved a minimum terrestrial distance of 600m.

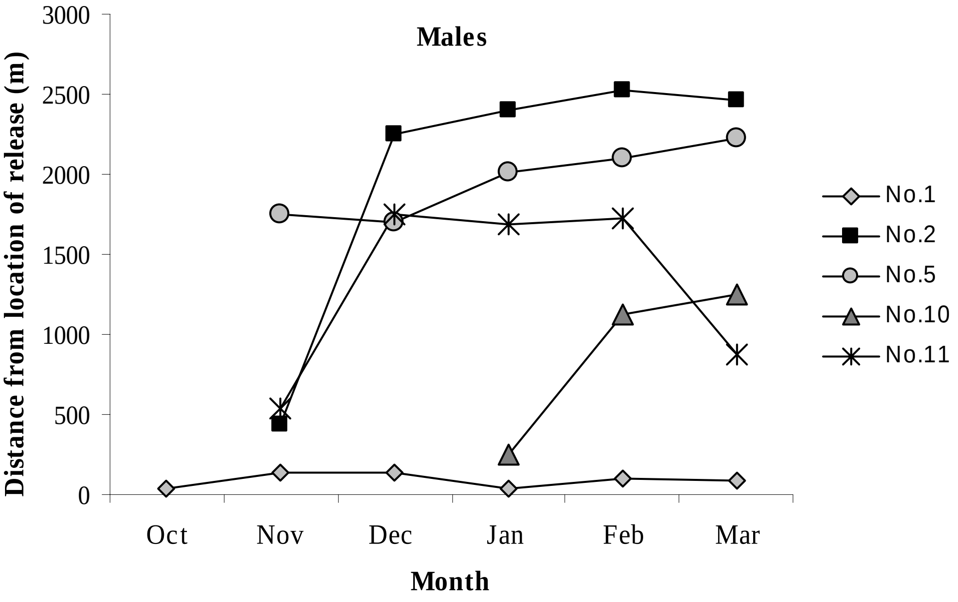
One terrestrial movement was recorded by spool-tracking. On 26 February, a female moved 700+ m from the southeastern end of the southern-most rice bay on the property, to the supply channel. This move was entirely terrestrial, occurred over a 26-hour period, and was associated with dry weather conditions.

Site Fidelity

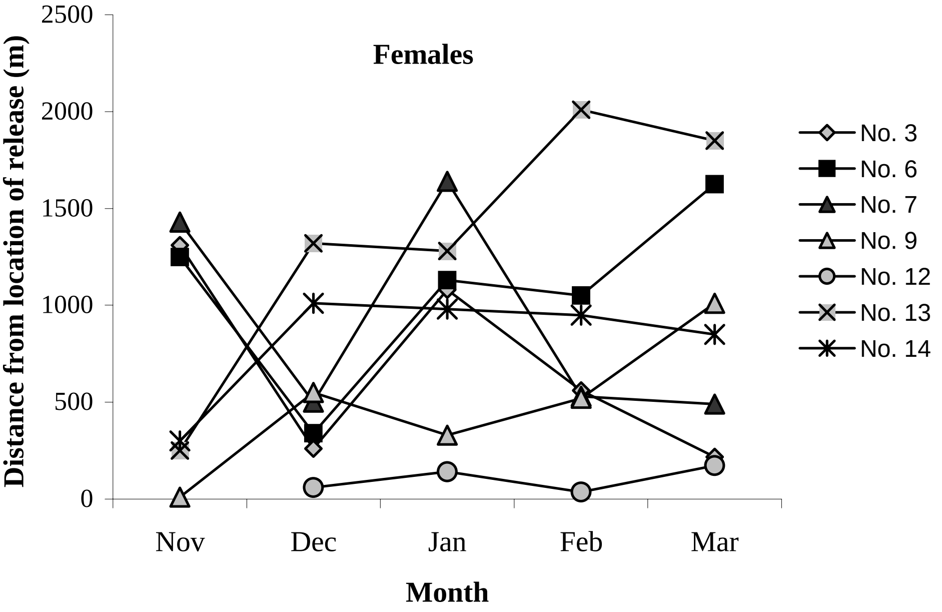
For all 12 radio-tagged turtles combined, there was no significant relationship between the distance from and time since their original release point (time expressed as the number of months that had passed since their release; F1,58 = 1.789, p = 0.187, r^2 = 0.039; Fig. 62). When regressed individually, this relationship was significant for only two turtles, number 5 and number 13 (Table 24, Fig. 62). The pattern displayed by the immature male (no.1) differed from the other males (all sexually mature). This male did not travel more than 200 m from his location of original capture, whilst all sexually mature males travelled at least 1 km from their location of original capture (Fig. 62).

Table 24. Relationship between the number of months since release and the distance travelled from location of release by individual *C. longicollis*. The relationship was examined using a linear regression, for the period September 2001 to March 2002. Two significant (*) regressions were found.

Turtle no.	**sex**	**F**	**P**	**R^2**	**n**
1	male	0.011	0.923	0.003	6
2	male	4.364	0.128	0.592	5
3	female	1.760	0.277	0.370	5
5	male	24.927	0.015*	0.893	5
6	female	0.963	0.399	0.243	5
7	female	1.081	0.375	0.265	5
9	female	8.163	0.065	0.731	5
10	male	5.333	0.260	0.842	3
11	male	0.101	0.771	0.033	5
12	female	0.589	0.523	0.227	5
13	female	11.704	0.042*	0.796	5
14	female	1.342	0.330	0.309	5



(a)



(b)

Figure 62. Relationship between the distance from, and time since (a) male and (b) female radio-tagged *C. longicollis* were relocated from their original release point, expressed in the number of months that had passed since their release.

Diet and Prey Availability

Stomach Contents

A total of 63 turtles were stomach flushed for prey items between November and late March, of which 40 were captured in rice habitats and 23 from irrigation channels. Eleven turtles (28 %) from rice habitats and 10 turtles (43%) from irrigation channels yielded no food (empty stomachs). Stomach contents revealed that turtles were feeding primarily on macroinvertebrates. However, turtles also consumed frogs, tadpoles, and fish.

Prey Availability: Abundance

The abundance of potential prey in all habitats sampled reached a seasonal peak in either December or January (Fig. 63). The dam habitats supported the highest mean abundance of invertebrates per sample (0.07 m²), with 1123 prey items/sample in the open section and 1303 items/sample in the vegetated section, followed by the rice tofo with a mean of 596 items/sample, the farm channel with a mean of 424 items/sample, the supply channel with a mean of 409 items/sample, and finally the rice field with a mean of 354 items/sample.

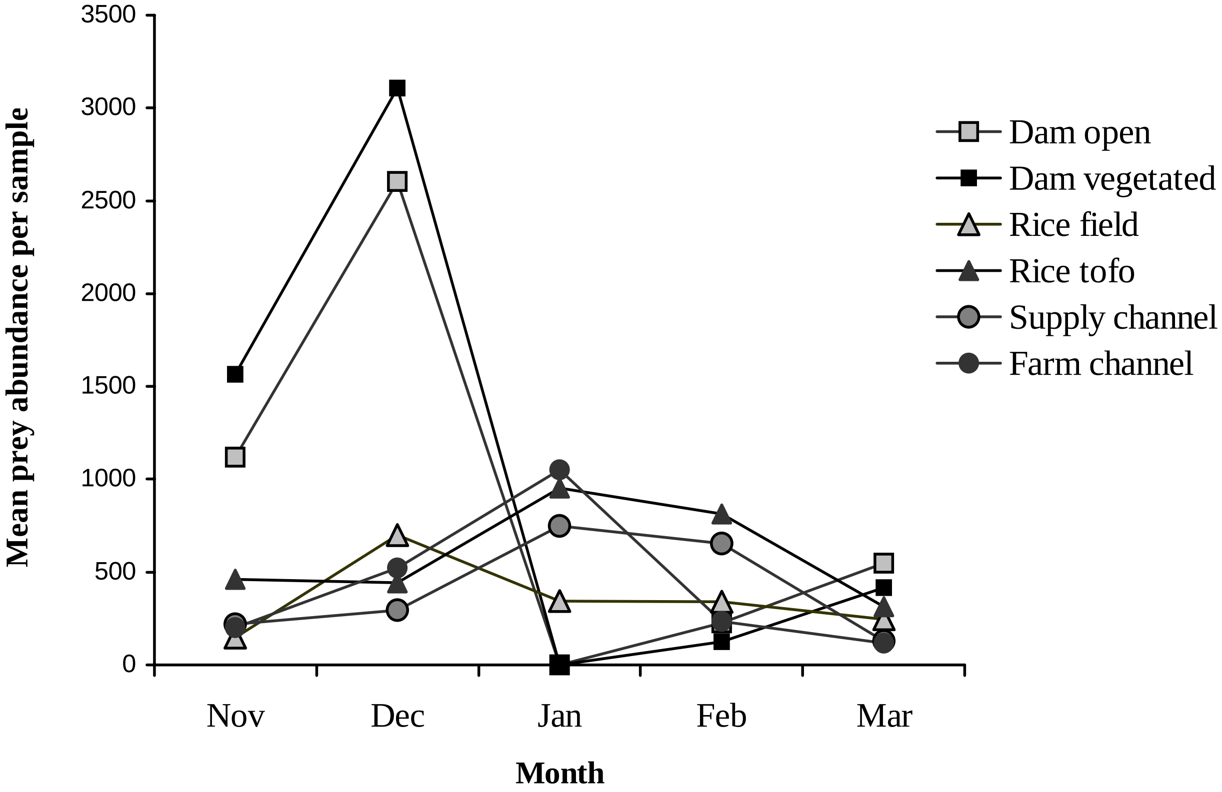


Figure 63. Pattern of prey abundance in each habitat type during the months of November 2001 to March 2002. Zero values for the dam in January are a result of the dam drying completely. Abundances for each month were derived from three column samples (area of 0.07m²) per habitat.

The abundance of prey was significantly influenced by month (F4,56 = 35.73, p < 0.0001) and by habitat type (F5,56 = 19.13, p < 0.0001). The effect of month on abundance was significant in all habitat types and the effect of habitat type on abundance was significant in all months (all p < 0.01). Prey abundance was consistently higher in the rice tofo than in the supply channel, differing by an average of 187 items/sample (Fig. 63). However, this difference was not significant in any month (all p > 0.05) except March, when the abundance in the rice tofo was significantly higher than in the supply channel (Bonferroni correction; p = 0.029). The rice habitats (combined) were not significantly different to the channel habitats (combined) in any month (F4,28 = 0.51, p = 0.617; all p > 0.05, Tukey's multiple comparisons).

The dam habitats were the most variable in terms of prey abundance (Fig. 63). The dam dried out during January; no samples could be collected in this month and prey abundance was considered to be zero. These zero values were not included in any analysis of either abundance or diversity, as this habitat was not considered available to turtles in this month.

Prey availability: Richness

The composition or richness of potential prey both within and among habitat types varied markedly over the study period. Dominant prey types overall included Chironomidae, Oligochaeta, Hemiptera, Coleoptera, Ephemeroptera, Odonata, Lymnophila, and Podocopida (Fig. 64; Fig.'s 63-67).

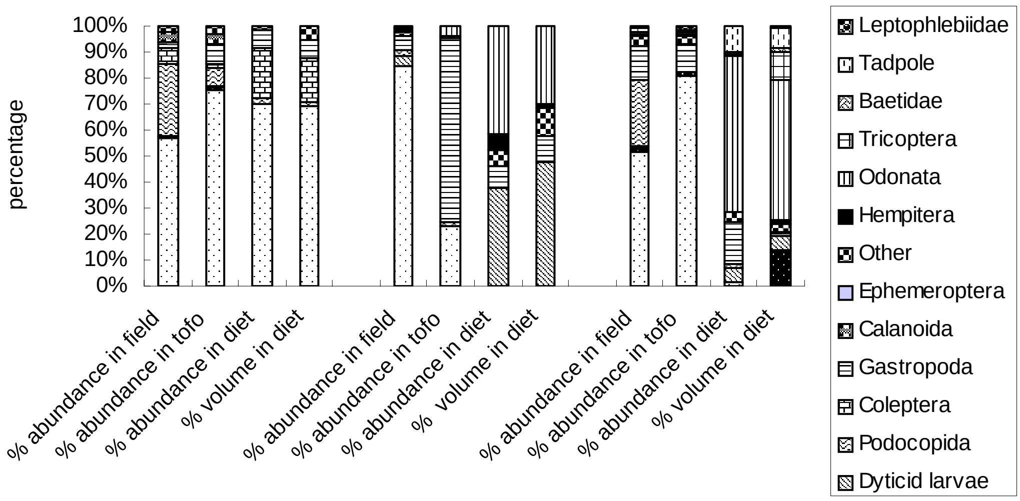


Figure 64. Comparison of stomach contents and available prey in the rice field and tofo habitats, showing chances in selectivity as prey richness increases.

Supply channel

The supply channel was dominated by chironomids, followed by ephemeropterans, oligochaetes and odnates (Fig. 65). There was a general trend towards an increase in the relative abundance of odonates, tricopterans and ephemeropterans from November to March. For example, odonates increased from a composition of 1% in November to 19% in March, tricopterans increased from 0% to 5%, and ephemeropterans increased from 6% to 38% over the same period. The relative abundance of chironomids was highest but variable, comprising 29%, in November, 65% in December, 42% in January, 61% in February, and 33% in March.

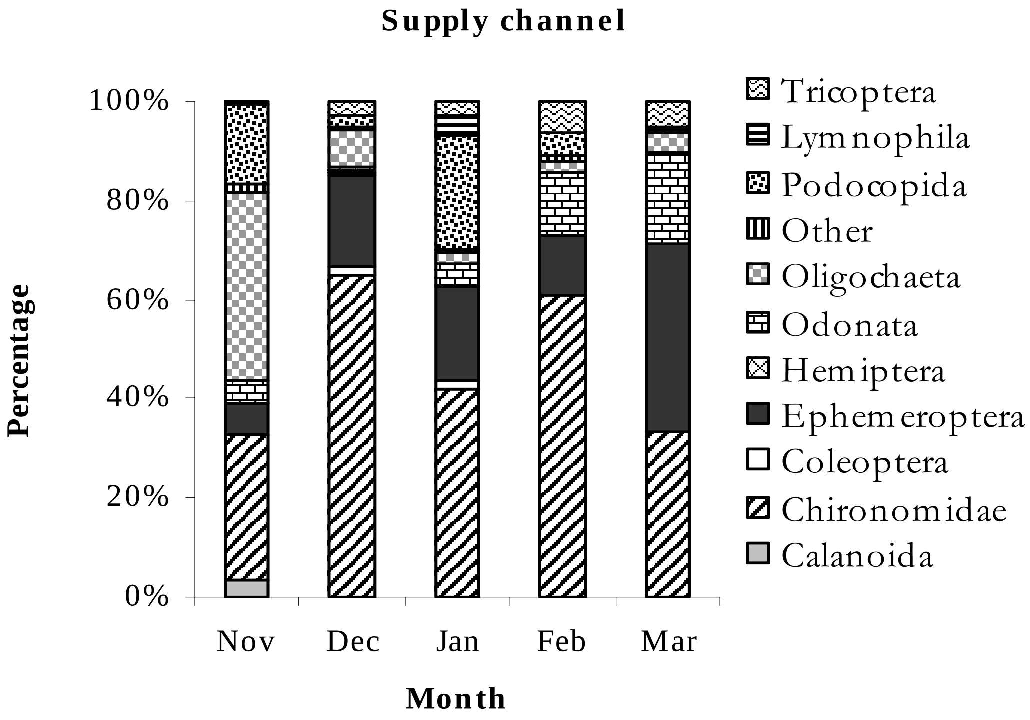


Figure 65. Richness or relative abundance of prey types in the supply channel from November 2001 to March 2002.

Farm channel

The prey composition of the farm channel was similar to the supply channel in that chironomids dominated (Fig. 66). However, farm channels differed from the supply channel in the relative abundance of some prey types, including ephemeropterans and odonates, which were less abundant in the farm channel. Also, both lymnophilans and hemipterans generally comprised a larger portion of the total abundance of prey in the farm channel than the supply channel.

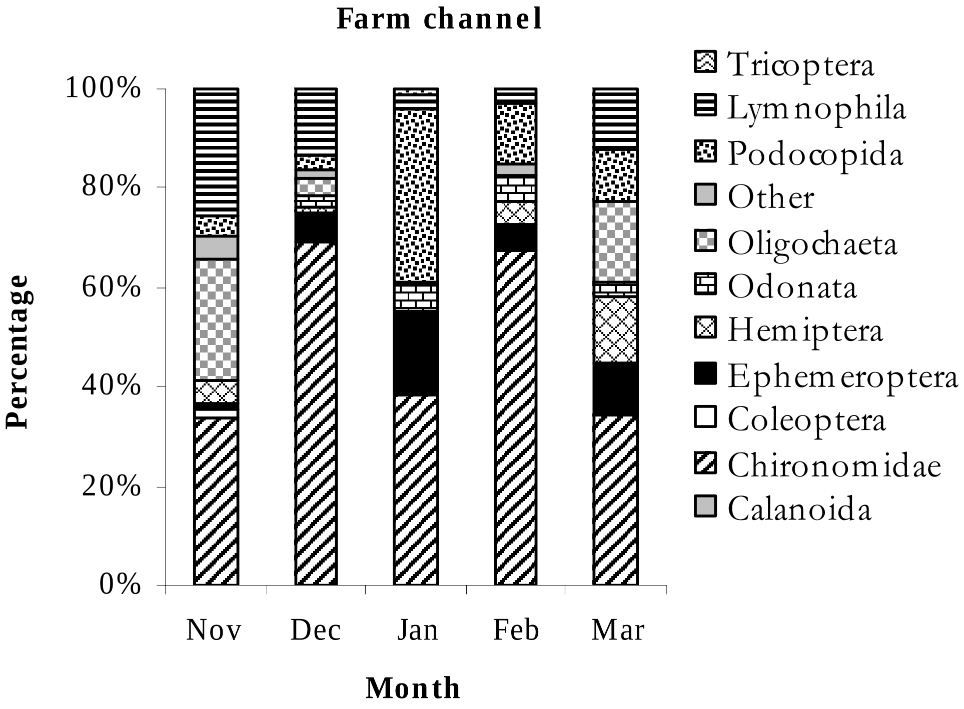


Figure 66. Richness or relative abundance of prey types in the farm channel from November 2001 to March 2002.

Rice habitats

The rice habitats were generally dominated by and chironomids and lymnophilans (Fig. 67; Fig. 68). Chironomids were more prevalent in the rice field, and the two groups were essentially in equal abundance in the rice tofo. Three new prey types colonised the rice tofo in December: odonates, ephemeropterans, and tricopterans. However, these prey types remained in relatively low abundance: by the end of the rice-growing period (late March) these groups comprised 1.5%, 3%, and 0.5% of the total abundance in the rice tofo respectively (Fig. 67). Hemipterans colonised both rice habitats beginning in February. In the rice tofo hemipterans comprised approximately less than 1% of the total composition in November, December and January but comprised 15% and 21% in February and March respectively (Fig. 67). Prey abundance did not differ between rice habitats (combined) and the channel habitats (combined) in any month ($F_{4,28} = 0.51$, $p = 0.617$).

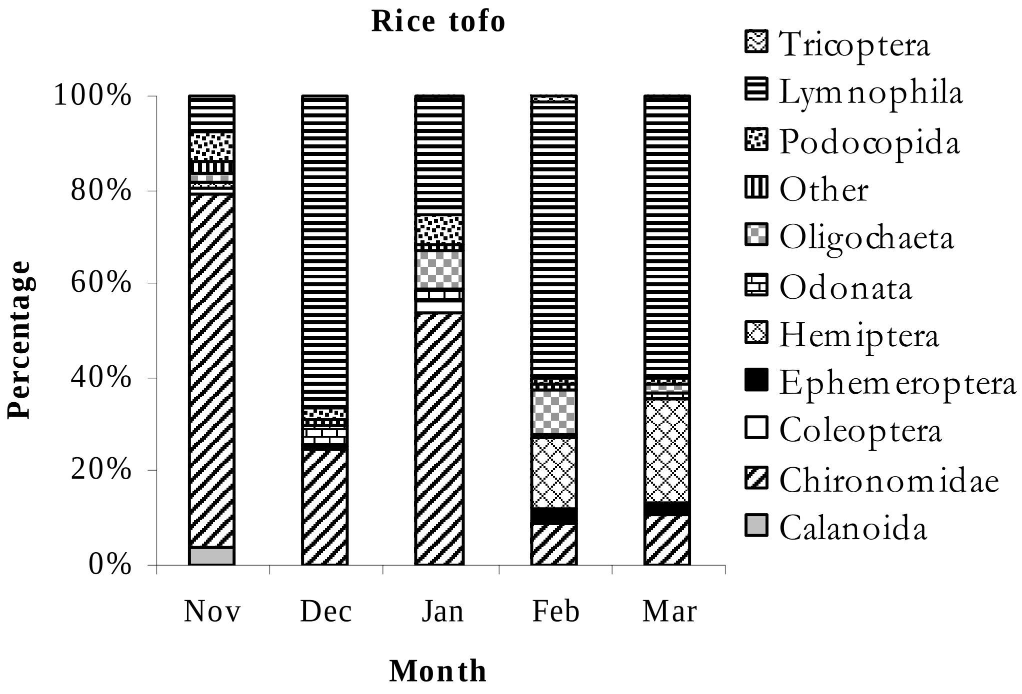


Figure 67. Richness or relative abundance of prey types in the rice tofo from November 2001 to March 2002.

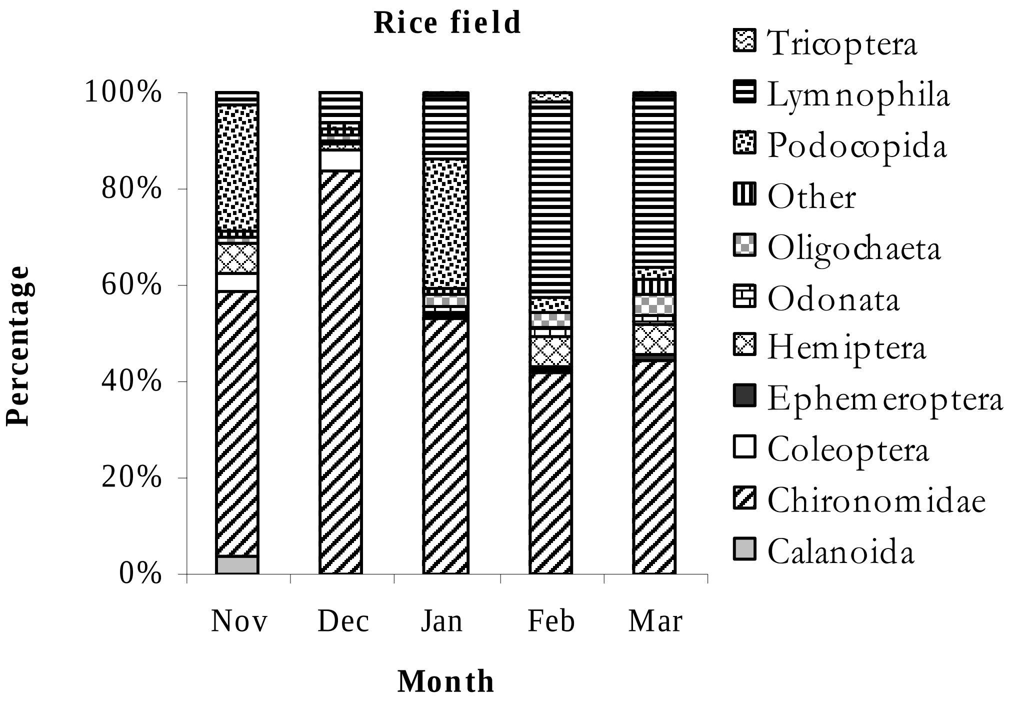
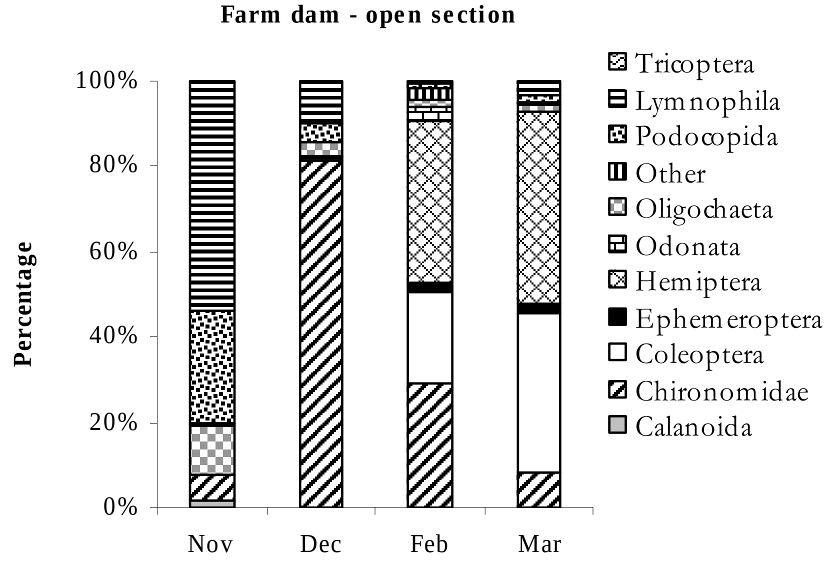


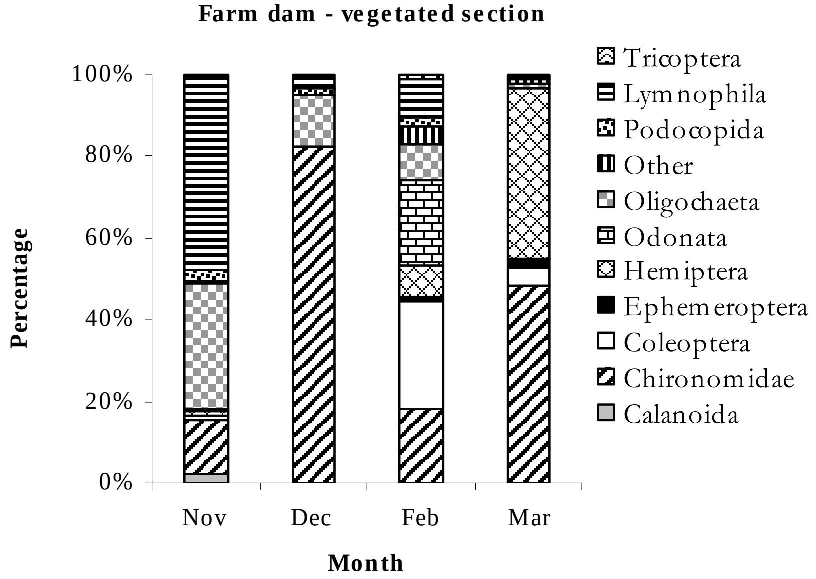
Figure 68. Relative abundance of prey types in the rice field from November 2001 to March 2002.

Farm dam

Both the open and vegetated sections of the dam were dominated by lymnophilans in November and by chironomids in December (Fig. 69a, b). The complete drying of the dam in January resulted in another shift in community composition. After re-filling (in February), there was a shift in community composition in the open section of the dam towards domination by hemipterans and coleopterans (Fig. 69a). This was less apparent in the vegetated section, where chironomids re-established a high relative abundance by March (Fig. 69b).



(a)



(b)

Figure 69. Richness or relative abundance of prey types in the open (a) and vegetated (b) sections of the farm dam from November 2001 to March 2002.

Prey availability: Diversity

Mean prey diversity over all months, expressed as Shannon's Diversity Index for families, was highest in the supply channel (1.58), followed by the farm channel (1.39), the rice tofo (1.13), the rice field (1.06), and the farm dam (open section = 0.96, vegetated section = 0.94). Family diversity (Shannon's Diversity Index) differed among habitats (F5,56 = 21.52, $p < 0.0001$) and among months (F4,56 = 11.05, $p < 0.001$) (Fig. 70). These two factors contributed 77% of the variation in diversity index. Family diversity was significantly different among habitats in all months except January (F3,56 = 0.86, p = 0.470, dam habitats excluded in January because the values were zero; Fig. 70). Month had a significant influence on family diversity in the rice tofo, rice field, and both sections of the dam (all $p < 0.05$), but did not significantly affect diversity in the supply channel (p = 0.515) or the farm channel (p = 0.282). Prey diversity was significantly greater in the supply channel than in the rice habitats (combined) (F5, 29 = 5.49, $p < 0.0001$), and a Tukey-Kramer Test revealed that this difference was significant in all months (all $p < 0.05$).

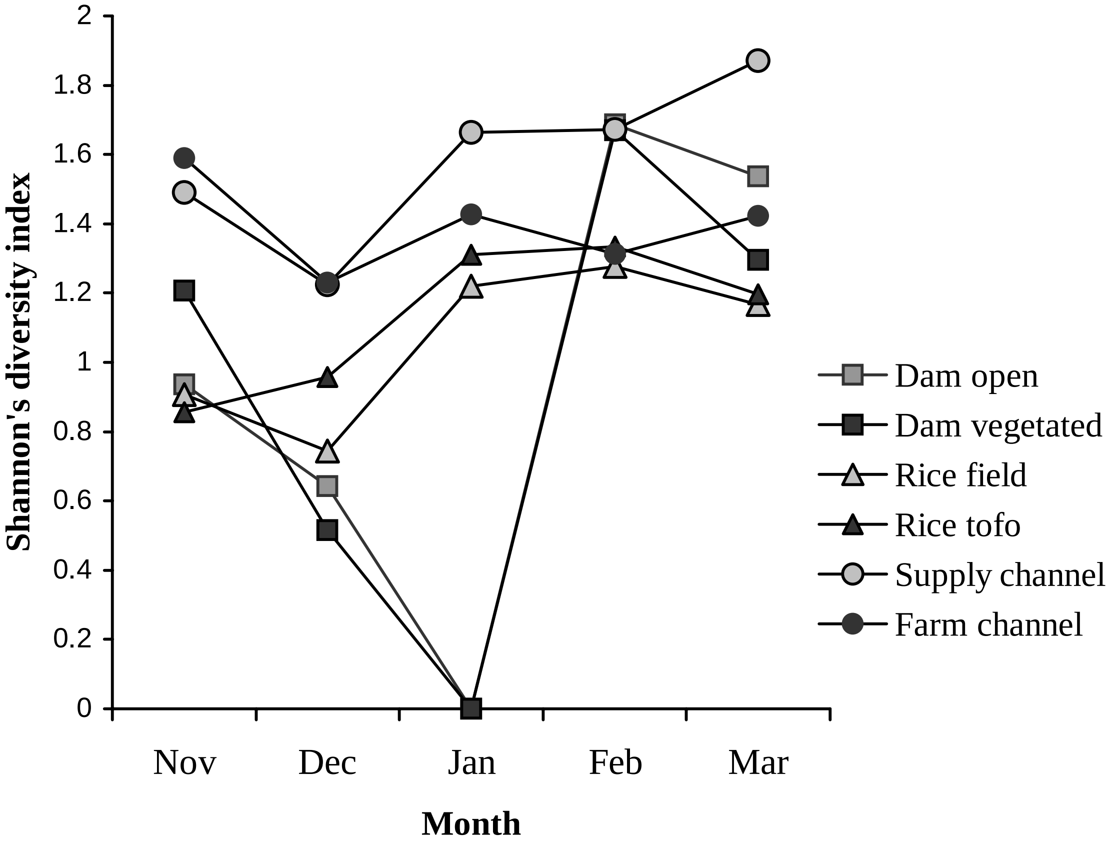


Figure 70. Pattern of prey diversity in each habitat type from November 2001 to March 2002. Diversity is presented as Shannon's diversity index, calculated at the family level. Diversity indices were derived from three column samples (area of 0.07 m²) per habitat. The zero values for the dam habitat in January are a result of the dam completely drying out in this month.

Comparing Diet and Prey Availability: Prey Selectivity

Visual examination of the data revealed that the richness of C. longicollis stomach contents from turtles captured in rice habitats corresponded closely with available prey in those habitats during the month of November (Fig. 64). This correspondence is supported by significant correlations between stomach contents and available prey for both rice habitats in November (Fig. 71a). However, in December and January the dietary richness did not correspond closely to prey availability (Fig. 64) and this was reflected in the lack of significant correlations between the two for both rice habitats in those two months (Fig. 71 b, c). A separate analysis, the Hotelling T2 Test, confirmed these results. The null hypothesis that all prey items in the rice habitats were equally preferred by turtles was rejected for December (F6,2 = 66.34, W = 4.34, p = 0.018) and for January (F6,2 = 243.08, W = 6.27, p = 0.039), but not for November (F6,2 = 1415.10, W = 15.01, p = 0.450).

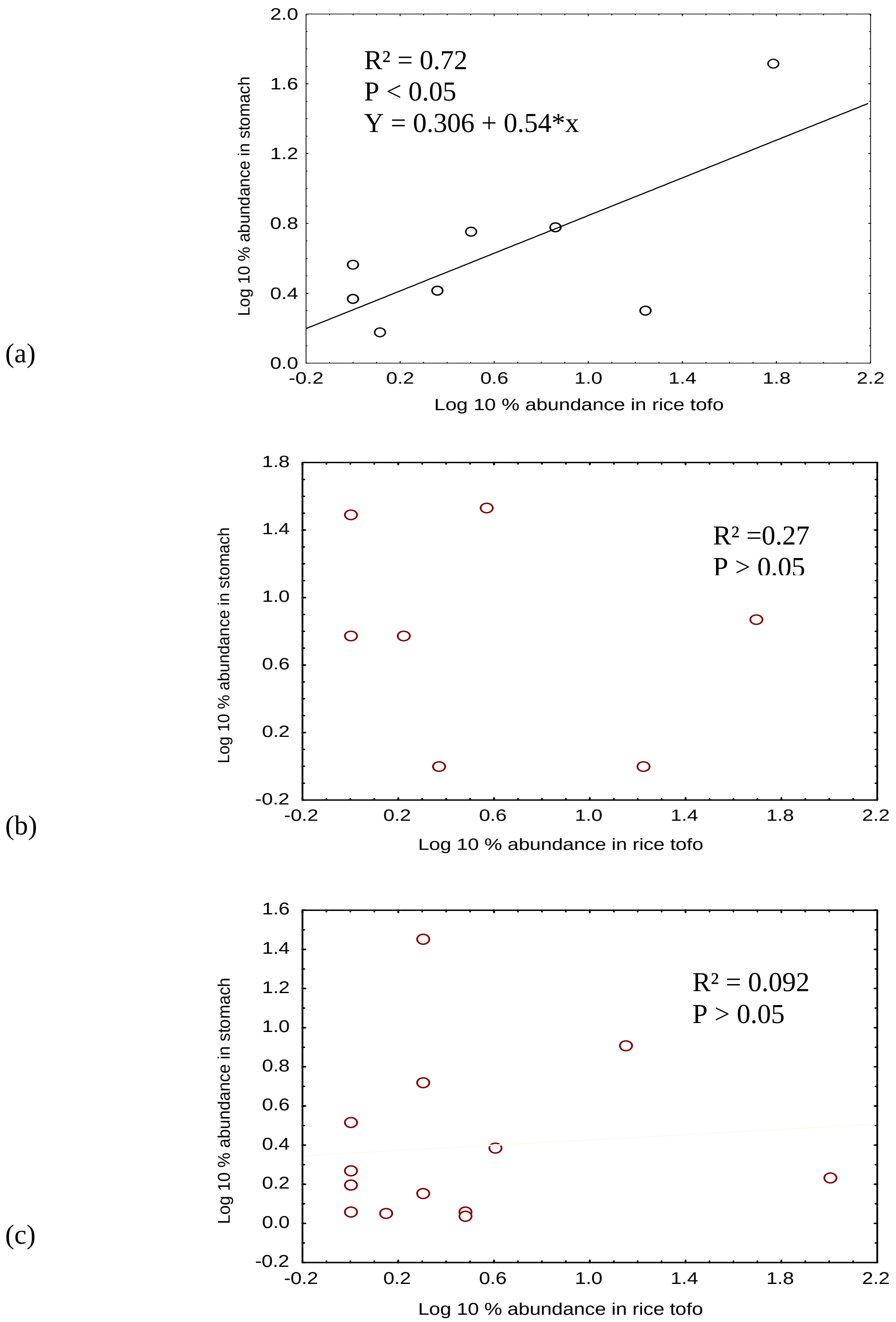


Figure 71. Relationship between stomach contents and prey availability in the rice. The relative abundance of prey types in the stomach contents of *C. longicollis* was significantly, positively related to the relative abundance of these prey types captured in the rice tofos in (a) November, but not in (b) December or (c) January. Values represent the logged percentage of total abundance of each prey type. The increase in prey selectivity in December and January relative to November was associated with an increase in prey abundance and diversity in the rice habitats

during this time (Fig. 63; Fig. 70). For example, prey diversity (rice tofos and rice fields combined) was significantly higher in January than in November ($F_{2,15}$ = 5.26, p = 0.019), and prey abundance in rice fields was significantly higher in December than in November ($F_{2,12}$ = 4.30, p = 0.039).

Discussion of Results

I. Vertebrate Diversity Utilising Rice Farms in the NSW Riverina

Our study supports the claim that there is considerable vertebrate biodiversity in the Riverina, despite the magnitude of habitat loss (Benson, 1999) and intensity of agriculture in the region (Australian Museum Business Service, 2000). Our study broadens this assertion into the context of on-farm vertebrate biodiversity. There have been few attempts at understanding how wildlife uses farms, and how to integrate that information into farm management practices; however, such efforts are increasing (Lindenmeyer et al., 2003). We conclude that the rice farm is an appropriate arena for management of biodiversity, although it should not be the only arena (biodiversity conservation in the region should be shared by as many stakeholders as possible).

Was Three Years Sufficient Time to Survey Most Vertebrates?

The use of species accumulation curves (SAC's) is invaluable in assessing biodiversity (Thompson and Withers, 2003; Thompson et al., 2003), provided that the surveys are not too small (Lande et al., 2000). In the present study, our SAC's indicate that we located the majority of species in each group during our 3-year study, given our techniques. Thus, we conclude that three years were sufficient to locate most vertebrates on rice farms in the region. Obviously, there will be species that we did not detect, due to their rareness, their secretive nature, or both. There will also be species that exist on farms, but not on the farms we sampled. A trade-off exists between number of sites and intensity of sampling. In the present study we chose to sample 10 farms intensively, because we wanted to say with confidence that we found the majority of species on those farms. A survey of more farms (e.g., 30) would be useful, especially if predictions of which vertebrates occupying those farms were made and tested.

Threatened Species and 'Significant Non-threatened' Species

We found several threatened species during the study, indicating that the rice agroecosystem in Australia, like most ecosystems, contains a subset of species in need of monitoring and/or management. This is in contrast to the unpublished opinions among some ecologists that the rice agroecosystem contains only generalist species that are widespread and abundant, and therefore requires little or no attention or management.

In particular, the fate of the southern bell frog, Litoria raniformis, may be partly in the hands of the rice industry. This species has experienced significant declines in recent years (Osborne, 1990; Ayers, 1995), possibly due to the introduction of an exotic fungus (Berger et al., 1999). Yet, a few populations remain, one of which resides on rice farms in the Coleambally Irrigation Area (pers. obs., Australian Museum Business Service, 2000; Pyke, 2002). On-farm management of this population could be critical to the survival of the population and even the species' persistence. A similar scenario exists in the western USA, where the persistence of an endangered species of snake, the giant garter snake (Thamnophis gigas) likely depends on its management along irrigation channels in rice-growing areas (Engles, 1994).

The finding of several threatened vertebrate species on farms highlights the need for significant species management in the Riverina. However, the industry's goal of 'prevention of further losses of biodiversity' (Mullins and Sutherland, 2001; Freudenberger and Stol, 2002) requires that we also consider species that are 'under threat' within the Riverina, regardless of their conservation status on state and federal levels. For example, extensive land clearing in the Riverina, although not conducted primarily for rice production, has fragmented woodlands, rendering some species uncommon or rare within the bioregion, despite their abundance elsewhere. However, the status of such species and how they use farms is often unknown. Examples of such species in the Riverina are the narrow-nosed planigale (Planigale tenuirostris; Fig. 72) and the inland carpet python (Morelia spilotes; Fig. 73). These two species currently have no official conservation listing, but appear to be uncommon, rare, or occur in isolated populations in the Riverina. Losing such species within the Riverina Bioregion (approx. 40,000 km2) would be a great loss to regional biodiversity, and would fail to meet the goals of the Australian Rice Biodiversity Strategy and Plan. Conversely, their retention would be beneficial to the overall biodiversity and therefore ecosystem health in the region, and to the species in general. Such conservation at a bioregional scale is consistent with Australian biodiversity strategies on state and federal levels (NSWNPWS, 1999; Environment Australia, 1996). A caveat is that the fate of such species in the Riverina should not be shouldered by rice growers alone (or any other single group – see Implications and Recommendations below).

Figure 72. A narrow-nosed planigale (*Planigale tenuirostris*) captured in a pit-fall trap line at Old Coree.

Figure 73. An inland carpet python (*Morelia spilota metcalfei*) captured by hand on Old Coree.

Comparisons to Current Knowledge and Previous Studies

Several issues make difficult any direct comparison of the vertebrate diversity between the present study and other studies in the Riverina Bioregion. First, our study was the first to target vertebrates exclusively on farms. In fact, because the most intact habitats occur at sites off farms, surveys often target off-farm habitats. Other factors hindering direct comparison are survey effort, timing of surveys, and location of surveys at various spatial scales. Birds are particularly frustrating in this respect due to their mobile nature.

Nevertheless, we have included the results of a few other studies for comparison. For example, we found 24 species of reptiles on the 10 farms, compared to 23 reptile species found at 33 sites in the Murrimbidgee Irrigation Area (MIA) (Sass et al., 2004), 24 reptile species found at 27 sites in the western slopes (Daly, 2004), and 13 species at 17 sites in the Coleambally Irrigation Area (CIA) (Australian Museum Business Service 2000). The MIA survey was conducted for one month in the summer of one year and included many sites with significant intact remnant vegetation (Sass et al., 2004), while the CIA study spanned spring and autumn of one year (Australian Museum Business Service 2000).

Similarly, in the present study we found seven species of frogs on the 10 farms, compared to 11 species found in a survey of 35 sites in the MIA (Wassens et. al., 2004), and five species found at 10 sites in the CIA (Australian Museum Business Service 2000). The CIA surveys were conducted during Spring and Autumn (Australian Museum Business Service 2000), while the MIA surveys were conducted in October (Wassens et al., 2004). Other surveys were conducted near the Riverina Bioregion (e.g., central western NSW, LeBreton et al., 2002; south-western slopes of NSW, Daly, 2004), and/or were restricted to certain groups (Sadlier and Pressey, 1994; Murphy, 1997;Webster et al., 2003).

Can Rice Farms Serve as a Surrogate for Natural Wetlands?

It has been suggested that rice farms may serve as surrogates for natural or semi-natural wetlands for some species or groups of species (Fasola and Ruiz, 1996; Fujioka and Lane, 1997; Elphick, 2000). On the other hand, in a general sense, irrigated systems have resulted in the degradation of wetlands and their biodiversity around the world (Gerakis and Kalburtji, 1998; reviewed in Lemly et al., 2000) and the general trend across agricultural landscapes is that, as farming intensity increases, biodiversity decreases (Wood et al., 2000; Donald et al., 2001). Our extensive study supports the idea that rice farms offer wet, feeding habitat for a suite of waterbirds, frogs, and snakes, and at least the latter two breed there. Our intensive study of how longneck turtles use rice farms also supports this idea, and a current study of the southern bell frog indicates the same (S. Wassens, Charles Sturt University, unpubl. data). It is likely that many vertebrate species thrive on irrigated rice farms, at least during the time when water is aplenty. However, it also likely that many species prefer natural wetlands over rice farms, due to differences in attributes such as vegetation structure, hydrology, prey availability, and refuge sites. Thus, the question is oversimplified. A better question may be, “which species utilise rice farms as surrogates for natural or semi-natural wetlands? Assuming that natural wetlands support more species than rice farms, in general (which needs to be demonstrated), a management extension of this question would be, “how can we enhance rice farms to attract more species”, or “how can we configure rice farms to support wetland species that are currently rare or non-existent on those farms?”

Influence of the Drought

Beginning in spring 2002 southeastern Australia was subjected to a severe drought (Bureau of Meteorology, 2002-3). This extended dry period resulted in extremely low capture rates of animals during the spring and summer of 2002-2003. During this time we decided to cease our standard pitfall trapping, as returns were zero to minimal. In addition, the drought influenced a specialised pitfall trapping effort to sample narrow-nosed planigales (Planigale tenuirostris) on Old Coree. We captured no planigales in the pitfall grid, which was designed to determine abundance and habitat use in the species (see methods). Narrow-nosed planigales likely aestivate during hot dry weather, when prey abundance is low. Low prey availability is known to induce torpor in this species (Dawson and Wolfers, 1978; Read, 1989).

Evaluation of the Techniques

General

The appropriateness, effectiveness, and efficiency of a given survey technique can vary considerably with landscape, site, and taxa targeted (Heyer et al., 1994). A sub-goal of our project was to evaluate our sampling techniques to facilitate further studies aiming to quantify vertebrate biodiversity on rice farms.

In general, most techniques we used were effective in detecting vertebrates on rice farms. Pitfall trapping resulted in the detection of several species that were found by no other sampling technique (e.g., narrow-nosed planigale, painted burrowing frog, two species of blind snakes), and was thus an important survey method. However, pitfall trapping did not detect species not found by other techniques on farms without remnant vegetation. Visual encounter surveys also produced species not detected by other methods (e.g., inland carpet python, curl snake, giant banjo frog). Bird counts were the obvious way of sampling bird species richness, and were used in tandem with bird call detection. Frog call surveys detected all frog species on farms without remnant vegetation, but not all frog species on farms with vegetation remnants (e.g., giant banjo frog, painted burrowing frog). Small mammal trapping (elliot traps) resulted in the detection of only one species (house mouse). Turtle trapping (hoop nets) was the best method of detection of long-necked turtles, but modified VES's (muddling in tofos of rice bays) also detected turtles. Bat trapping (harp trap), although not used extensively in the study, was effective in detecting four species of bats not detected by any other method (however, we recommend using automated call recording for future bat surveys (ANABAT – Johnson et al., 2002).

In summary, we recommend using all of the above techniques to determine species richness on rice farms. However, answering more refined questions concerning biodiversity on rice farms might require using only one or a few techniques. For example, pitfall trapping would probably be sufficient for comparing frog abundance among rice farms, while frog call surveys might suffice for determining frog species richness. Our surveys had the general aim of determining what species occupy rice farms, which we felt required the use of several techniques.

Questionnaire Surveys: Involving the Farmer

The questionnaire can be a powerful survey method for understanding a farmer's knowledge, attitude, and intentions toward biodiversity conservation (e.g., Jacobson et al., 2003). Our 'significant species questionnaire' for farmers was a pilot study designed to reveal the potential effectiveness of such surveys for preliminary assessments of the presence-absence of on-farm species. In particular, the survey takes advantage of a farmer's accumulated observations of animals on his/her farm. In this way the farmer survey can be used as a surrogate or pilot for actual field surveys conducted by researchers, the latter being not only more resource-expensive, but prohibitive on most larger scales. In practice, such a questionnaire could greatly reduce the costs of determining the presence of certain species at local scales by creating a starting point for field surveys.

Although these questionnaires have limitations such as errors in species identification and lack of knowledge of secretive species not readily seen, we argue that when used carefully the surveys can be a powerful tool for determining presence-absence of some species. The critical issues are (1) obtaining large sample sizes to overcome errors such as species identification; and (2) choosing species that can readily be distinguished from other species by farmers.

For example, our one page questionnaire included sharp colour photographs of four significant species (Appendix 2). However, it came to our attention that many farmers could not distinguish between the narrow-nosed planigale (one of the four animals) and the common house mouse. Indeed the differences are subtle. Thus, the narrow-nosed planigale is probably not a good species for such a survey. In contrast, the inland carpet python is a good candidate for questionnaires because it is both easily distinguished from other snake species due to colouration and girth, and is well-known among the target group - farmers in the Riverina.

The southern bell frog perhaps illustrates best the potential utility of the farmer survey. This is because the species is easily identifiable (the only large green frog in the area), and because we can ground truth our survey because we know areas where the species occurs. The southern bell frog was once widespread in southeastern Australia, but has undergone severe declines in much of its range (Osborne, 1990; Ayers, 1995), possibly due to the exotic fungus that threatens many frog species in Australian and worldwide (Berger et al., 1995). The species is known to utilise rice bays (Ehmann and White, 1996), and now exists in isolates, one of which centres around the town of Coleambally (Australian Museum Business Service, 2000). However, the geographic limits of the isolate are poorly known (pers. obs., Australian Museum Business Service, 2000; Pyke, 2002), and so a survey of farmers in the area could potentially lead to the discovery (and therefore, management) of nearby populations. Fortunately, farmers that participated in our survey owned properties in and around Coleambally (Fig. 7). Thus, we knew it was likely that farmers from Coleambally claiming to have seen southern bell frogs on their property had actually seen the species (rather than a misidentification). This apparently resulted in a high occurrence rate for southern bell frogs among Coleambally farmers (12 of 18 = 67%). We could argue, therefore, that similar occurrence rates reported by farmers from other areas (where populations of the species may persist) could reflect actual populations, rather than represent species misidentification – provided that the sample size is high enough.

Examining our survey results, we find that one other area with > 10 participating farmers shows a relatively high occurrence rate (Jerilderie, 4 of 10 = 40% reporting having seen the species; Fig. 7). The Jerilderie area is within 70 km of Coleambally, and may well contain populations of southern bell frogs. Although a larger sample than that participating in our survey is needed, we feel that a larger sample would result in the responses falling out into two groups: (1) low occurrence rates = misidentifications; and (2) high occurrence rates = actual populations. Examples of low occurrence rates of southern bell frogs in our survey that probably represent misidentifications are: Finley, 3 of 14 = 21%; Deniliquin, 2 of 15 = 13%; and Leeton, 1 of 8 = 13% (Table 7). In summary, a questionnaire targetting a larger sample of farmers in the area would be a valuable, cost-effective way to discover remnant populations of southern bell frogs, especially given the costs of researchers detecting the species on such a large sample of farms.

II. Factors Underpinning Vertebrate Diversity on Rice Farms

Our studies indicated that there are two major factors underpinning vertebrate biodiversity on rice farms: productive rice bays and remnant vegetation patches (Doody and Osborne, 2003; Doody et al., 2004). We acknowledge that other drivers of biodiversity likely exist on rice farms (e.g., presence of feral animals, grazing regime), but our data implicate rice bays and vegetation remnants as two major drivers.

Productive Rice Bays

Rice Bays and Frogs, Snakes, Turtles, and Aquatic Birds

A serendipitous weather pattern in 2001 and captures of large numbers of young frogs dispersing from rice bays allowed us to make a crude estimate of annual frog production in Riverina rice bays (spotted grass frogs only). March and early April 2001 were very dry, with little or no measurable rain at the study farms. During this time the rice crops were harvested and the bays drained, leaving a small amount of water in the bays, along with rice crop stubble. Apparently, during this time, many or all of the spotted grass frog tadpoles had transformed into frogs, and were waiting for significant rainfall to disperse from the bays.

On the day and night of 21 April, area farms received heavy rainfall (27 mm at Old Coree, Russell Ford, Rice Research, unpubl. data) at a time when we had begun to install pitfall trap-lines. The next day one of us (SD) discovered great magnitudes of young frogs (mostly spotted grass frogs) in the pitfall buckets, particularly along one trap-line at Old Coree. A total of 1631, > 95% yearling, spotted grass frogs (Limnodynastes tasmaniensis) were captured in the trap-line buckets at the Old Coree trap-line during the night of 21 April. Only two of these frogs were in buckets on the remnant side of the rice bay side of the trap-line, indicating that the frogs had dispersed from the rice bays.

This was the only trap-line at Old Coree that was adjacent to rice bays (approx. 10 m away from, and parallel to the closest bay). On the other side of the trap-line was riparian remnant vegetation (grazed, open eucalypt woodland with a few saltbush along a large 20 m wide slow flowing Billabong Creek). We hypothesized that this was a major dispersal event for the frogs for the year, though it was likely not the only dispersal event, as spotted grass frogs have an extended breeding season (Barker et al., 1995). Assuming that the frog production of these rice bays was typical or average, we estimated the number of frogs produced annually in NSW Riverina rice bays to be in the order of 500 million. This number could be conservative, as spotted grass frogs have an extended breeding season that may encompass several dispersal events per year.

This observation of mass dispersal of frogs leads us to hypothesize that frogs represent the highest biomass of any vertebrate group on rice farms, and that spotted grass frogs may comprise the highest biomass of any species inhabiting rice farms. This finding has implications for food web dynamics and the importance of frogs in the rice agroecosystem. Because frogs are a staple for many vertebrate species, including many snakes, birds, and mammals, they are likely a key energy link in the rice agroecosystem. One finding highlighting the importance of frogs in the food web is the observation of brownsnakes feeding on frogs despite their reputation as mainly lizard and mammal feeders (Wilson and Knowles, 1988; Shine, 1991; Cogger, 2000; but see Phillips and Fitzgerald, 2004). Brownsnakes were common on rice farms, utilising areas in and around irrigation channels and rice bays (see also Whitaker and Shine, 1999a; 1999b; 2000; 2003).

It follows that maintaining a healthy frog biota on rice farms would be important in maintaining current levels of frog-eating predators such as snakes and aquatic birds. Three areas need further research here. First, a study should attempt to confirm that frogs indeed represent the highest biomass of any vertebrate group on rice farms. Second, frog production in irrigation channels and dams would complement our findings by extending them to the entire rice farm. Third, a study of the food web in and around rice farms would better indicate how much of the frog production is utilised by the next trophic level. Such studies would more accurately highlight the importance of frogs in the rice agroecosystem.

We also quantified use of rice bays by large numbers of waterbirds, including ducks, ibis, herons, and egrets. Irrigation channels were less frequently used by these birds, which presumably exploited the shallow water in rice bays. Waterbirds have probably coped with landscape changes associated with farming better than most other inland fauna (Briggs, 1994).

Eastern long-necked turtles (Chelodina longicollis) were common on a few farms, rare on others, and seemingly absent on others still. Where common they utilised the more permanent irrigation channels during autumn (and presumably winter), but many moved into seasonally-available rice bays during spring and summer. Turtles moved into flooded rice bays soon after flooding, utilising mainly the tofos around the bays, but occasionally foraging in the rice crop itself. A similar pattern of habitat utilisation has been found for the southern bell frog (Litoria raniformis). Radio-tracked frogs were observed to use irrigation channels in November, but moved into rice bays in January (S. Wassens, Charles Sturt University, unpubl. data).

Finally, snakes, although few in species, were extremely abundant on some of the farms. In particular, eastern brownsnakes and tiger snakes inhabited vegetation in two areas: the rows separating the rice bays, and along the irrigation channels. Red-bellied blacksnakes also utilised those areas, but were more often found in or near remnant vegetation patches. Lizards were not particularly common in and around rice bays, nor were mammals, with the exception of house mice and foxes.

Rice Farms vs. Dryland Cropping: More or Less Biodiversity?

The present study was not designed to directly compare vertebrate biodiversity between rice farms and dryland farms – indeed, surveys were restricted to rice farms. However, within rice farms we sampled habitats that could be used as surrogates for such a comparison. We employed pitfall trap-lines along rice bays and along paddocks of canola and fallow/stubble, allowing a comparison of relative abundance and richness of certain vertebrates between the rice and ‘dry’ crops. We also conducted pre- and post-flooding (irrigating) bird counts on each farm, which would also be indicative of a rice farm-dryland farm comparison.

Capture rates of frogs were 1.5-2 times higher along rice bays than along dry crops, although the difference was statistically significant only in spotted grass frogs, due to low sample size and high within-group variance (Table 12). Other vertebrates captured in pitfalls showed no differences (eastern brownsnake, house mouse, boulenger's skink, all skinks, all lizards, all reptiles; Table 12). In a similar comparison of wet vs. dry habitats, which also included dams and black box depressions, frogs were also much more abundant in wet habitats (Fig. 26; Table 11).

Species richness did not differ between rice and dry crops when considering small animals captured in pitfall traps. However, pre- and post-flooding bird counts demonstrated that flooded rice bays were used by more species and more individuals than dry rice bays, most of which were aquatic or semi-aquatic species (Fig. 28, Fig. 29). Thus, given all else equal, rice farms are likely to contain higher abundances of frogs and birds, and higher species richness of birds, than dryland farms. However, this assertion needs to be tested by comparing richness and abundance between rice farms and dryland farms.

Remnant Vegetation Patches

Both the extensive study and the 'birds and remnants' study strongly supported the importance of remnant vegetation patches to on-farm vertebrate wildlife. Although the number of farms sampled was modest, the extensive study revealed that farms with vegetation remnants (ranging from black box depressions to woodlands to sandhills) contained more species of vertebrate animals than those without remnants. In particular, woodland birds and lizards were considerably more diverse on farms with remnants (Fig. 23; Fig. 24). However, all groups exhibited higher species richness on farms with vegetation remnants (Fig. 22; Fig. 24). While this finding seems intuitive, it provides a real foundation for on-farm management on Riverina rice farms.

Lizards are apparently the group that has coped best with fragmentation of habitat in Australian agroecosystems (Kitchener et al., 1980; Burbidge and McKenzie, 1989; Smith et al., 1996), apparently because of their low energy and/or space requirements compared to mammals and birds (Turner et al., 1969; Nagy, 1987). We found higher richness and abundance of lizards on farms with remnant vegetation patches (Fig. 24; Fig. 25). Although not an objective of the present study, it is likely that the condition of remnants influenced lizard richness and abundance. The number of trees positively influenced gecko abundance (Smith et al., 1996), and area of woodlands positively influenced lizard richness (Kitchener et al., 1980; Kitchener and How, 1982; Smith et al., 1996) in the wheat-sheep belt of Western Australia. Some of our on-farm remnants in the extensive survey were very small (e.g., a few hectares), grazed, and exhibited no regeneration. Yet these remnants contained up to five lizard species, supporting the notion that even the smallest and most degraded remnants have some conservation value (Smith et al., 1996). However, our pseudocontrol site, which encompasses more habitat types than the other farms, contained more then double the number of lizard species (Table 8). Thus, although lizards in the Riverina may be more resilient to habitat fragmentation than mammals or birds, there is likely still a significant impact of remnant size on their prevalence there.

The 'birds and remnants' study, conducted solely in Black Box remnants, indicated that the size of remnants was also critical for some bird species. Woodland and DECREASER bird species richness were both positively influenced by patch area (Table 18; Fig. 35; Fig. 36). Woodland bird abundance was also influenced by patch area, although the difference only approached statistical significance (Table 18). In addition to area, the condition of the remnants influenced bird richness and abundance (Fig. 43; Fig. 44). In other words, remnants with more understory vegetation harboured more bird diversity.

How large of a patch is needed to harbour considerable biodiversity? At present, the amount of habitat needed by a particular species or group is poorly understood (Lindenmeyer et al., 2003), and would be expected to vary depending on patch type and landscape variables such as the distance from the nearest other patch (e.g., Lindenmeyer and Peakall, 2000). Our results indicate that for woodland bird species on rice farms ‘bigger is better’, when considering the range of patch sizes from 2 to >50 hectares, while most ‘declining species’ (sensu Reid, 1999) inhabit patches > 50 hectares in size (Fig. 30). The target group will therefore dictate any management practices initiated to enhance biodiversity on and off farm.

Although we did not examine the influence of vegetation remnant condition on diversity and abundance of other animal groups, there are likely to be patterns. For example, certain species of skinks prefer core areas of remnants to edges (Anderson and Burgin, 2002), and the presence of logs, rocks, bark, and leaves enhances small vertebrate populations (Lindenmeyer et al., 2002; Lindenmeyer et al., 2003; Fischer et al., 2004). Grazing regimes that cause major structural change to the environment (Saunders et al., 2003) can disadvantage some species of vertebrates (reviewed in James, 2003). Maintaining some grazing-free areas on properties could be important to the persistence of populations of certain species, especially during times of environmental stress (James, 2003). For example, the skinks Menetia greyi and Morethia boulengeri, both of which were found in the present study, were found to be more common on sites with little or low intensity grazing (Henle, 1989; James, 2003).

Conclusions: What Would a Biodiverse Rice Farm Look Like?

Biodiversity refers to all species of plants, animals, and micro-organisms existing and interacting within an ecosystem (Vandermeer and Perfecto, 1995). Biodiversity has two readily measurable components: richness or the number of species, and abundance. Thus, a particularly biodiverse rice farm would have high numbers of lots of species. Interestingly, our study found that the two major attributes underpinning on-farm biodiversity – productive rice bays and remnant vegetation patches – influenced not only different groups of animals, but also influenced those groups in different ways. Productive rice bays seemed to have had a profound influence on abundance, while remnant vegetation patches generally boosted species richness.

Based on the above, theoretically the most biodiverse rice farm might be one that has the best mixture of rice bays and native vegetation. For example, farms with more remnant vegetation will contain more woodland bird species, more species of mammal, more lizard species, more species of frogs, and more snake species, while farms with rice bays will contain a higher abundance of frogs, snakes, turtles, and aquatic birds, and more species of aquatic birds (Table 25). Unfortunately, there is no exact recipe for improving farm biodiversity – every farm will be slightly different than the next (Salt et al., 2004), and it would take decades or longer to unravel all of the complexities for each area (i.e., genetic diversity, food web dynamics, energetics, habitat utilisation). We can, however, use the wealth of information on biodiversity in agroecosystems that is emerging, to formulate a sensible, practical, cost-effective plan for improving biodiversity on farm. The present study indicates that such a plan should include the improvement and revegetation of habitat remnants among rice bays and dry paddocks (see next section).

Table 25. Summary of the comparative influences of rice bays and vegetation remnants on components of biodiversity of different groups.

rice bays	vegetation remnants
aquatic bird richness and abundance	woodland bird richness and abundance
frog abundance	frog richness
snake abundance	lizard richness
turtle abundance	snake richness
	mammal richness

III. Remnant Vegetation Patches, Bird Diversity, and Management Implications

Bird Fauna and Composition

The number of species that were recorded during the present study (115 species, Appendix 3) is similar to that that has been recorded during other short-term bird studies within woodland areas of southeastern Australia. Eighty-seven species were recorded in Bull Oak remnants of western Victoria, VIC (Watson et al., 2000), 86 in woodlands of the Northern Tablelands, NSW (Barrett et al., 1994), 115 in woodland/ forest in northern NSW (Oliver et al., 1999), and 94 in woodland of central NSW (Seddon et al., 2001). Therefore, despite the Coleambally Irrigation Area (CIA) having only 3.2% woody vegetation cover, it still supports a relatively diverse bird fauna. Of the 115 species recorded during the present study, 72 were utilising the Black Box habitat within the survey transects (Appendix 4).

Of the 21 species that occurred across most sites (Table 13), the majority were “INCREASER” species. INCREASER species are habitat generalists that can exploit the matrix habitat better than habitat specialists, and are thus more widely distributed across agricultural landscapes (Reid, 1999) (Appendix 4). INCREASER species also contributed a higher number of individuals per species group (density) than the other species groups (Fig. 34). This is likely to be due to the fact that INCREASER species are often gregarious in social organization (Reid, 1999).

The CIA also offered important habitat for some species that have been found to be DECREASERS in the sheep-wheat belt (Reid, 1999). Groups such as the small insectivorous passerine group that form the bulk of the DECREASER species (Reid, 1999) are relatively well represented in the CIA. DECREASER species such as the chestnut-rumped thornbill (Acanthiza uropygialis), Jacky Winter (Microeca leucophaea), red-capped robin (Petroica goodenovii), rufous whistler (Pachycephala rufiventris) and Restless Flycatcher (Myiagra inquieta) were present, but species such as the brown treecreeper (Climacteris picumnus), southern whiteface (Aphelocephala leucopsis), hooded robin (Melanodryas cucullata) and diamond firetail (Stagonopleura guttata) were not detected during the present surveys. We also documented a nationally threatened DECREASER species, the superb parrot (Polytelis swainsonii). The CIA region apparently offers feeding resources for this species. In spring, flocks of up to 46 individuals were observed feeding on Acacia spp. seed (D. Bourne, pers. obs.), and in autumn the species has been observed to feed on spilled rice in the region (Australian Museum Business Service, 2000).

Habitat and Landscape Variables Across Sites and Size Classes

There was a general lack of patterns in landscape and habitat variables across size classes (Table 15). The potentially confounded factors of total shrub cover and habitat complexity were not related to patch area (Table 15). These variables have, in some cases, been found to significantly influence species richness and the prevalence of certain bird species (Freudenberger, 1999; 2001; Seddon et al., 2001). Exceptions were: (1) the number of dead trees and stem density decreased with increasing patch size class (Table 15); and (2) the number of patches >10 ha within a 2 km radius, log volume, and the height of the canopy all increased with increasing size class (Table 15).

In the present study a Principal Component Analysis (PCA) was conducted to show separation between sites based on a sub-set of landscape and habitat variables. The pseudocontrol site (site 210) proved to be distinct from other sites based on the PCA ordination (Figure 30). Site 210 also contained a bird assemblage (Group 1) that was distinct from other sites (Figures 31, Fig. 32a, b). Sites 179, 71, 12 and 15 were also distinct in the PCA, of which the latter three were also included in a group with a distinctive bird assemblage (Group 2) from the “bulk” of other sites (Group 3) (Fig. 31; Fig. 32a, b). Site 192 was characterised by extensive eucalypt dieback, high geographic isolation and was shown to be distinct in the PCA (Fig. 30). However, its bird assemblage was not different from that of Group 3 (Fig. 32a, b).

Due to the low explanatory power of the PCA ordination (Fig. 30), the principal components (PC’s) were not considered in any further analysis. This low explanatory power could be partially explained by low sample size of the present study (n = 23). Comrey and Lee (1992) suggest that a sample size of 100 is poor and 50 very poor for a PCA. Another reason for the poor explanatory power of the PCA may be the lack of variability in some of the variables included in the analysis (Table 16). A lack of variability in the data can lead to poor correlations between variables and thus a poor explanatory relationship for the PC’s since a PCA relies on the degree of correlation between variables to compose PC’s (Tabachnick and Fidell, 1996).

Assemblage Similarities Across Sites and Size Classes

A significant area effect was observed in bird assemblage composition. Based on logged mean abundance of bird counts, three distinct groups of assemblage structure were observed (Fig. 31; Fig. 32a, b). Group 1 consisted of Site 210 that was the pseudocontrol for the study and was most distinct from all other sites (Fig. 31). Group 2 was distinct from Group 3 at a similarity of slightly under 40% (Figure 31) and consisted of small and very small remnants (<2.5 ha) that were predominately heavily degraded sites (Fig. 32a, b). The remainder of the sites composed Group 3, which formed the bulk of the study sites (Fig. 32a, b).

Individual species prevalence across sites was distinct in the pseudocontrol site, which affected the overall composition of the bird assemblage, and resulted in distinct groupings based on the assemblage structure (Fig. 31; Fig. 32a, b) (Appendix 4). Differences observed in bird assemblage structure in fragmented landscapes have been attributed to the fact that different components of an assemblage react differently to the effects of fragmentation (Bennett and Ford, 1997; Ford et al., 2001; Major et al., 2001; Seddon et al., 2001). Differences in responses to fragmentation have been largely related to the feeding guild of the species or species group (Ford et al., 2001; Major et al., 2001; Seddon et al,. 2001). Within southeastern Australia small, insectivorous species have been found to be the species group that reacts in the most sensitive manner to fragmentation (Bennett and Ford, 1997; Major et al., 2001; Seddon et al., 2001). Watson et al., (2000) found that insectivorous species still inhabited small and relatively degraded remnants of Buloke (Allocasuarina luehmanni). The most likely cause for their finding was the lack of interspecific competition from the noisy miner (Manorina melanocephala) that was only present in the area in low densities (Watson et al.,2000). Some species groups, including some granivores and nectarivores such as the noisy miner, have benefited in most cases, from fragmentation (Seddon et al., 2001). These habitat generalists or “matrix species” are ubiquitous within highly fragmented landscapes (Major et al., 2001; Seddon et al., 2001).

Habitat Variables and Birds

Patch Area

Patch area and species richness

Many studies have demonstrated a positive relationship between species richness and area, and this relationship forms one of the bases of the theory of island biogeography (MacArthur and Wilson, 1967). Three main causes have been implicated as contributing to the species-area relationships: (1) random sampling effects, larger areas contain more individuals and thus, by chance alone, will contain more species; (2) habitat diversity, whereby larger areas contain more diverse habitat, thus offering more niches to be exploited; and (3) extinction-colonisation concept, whereby larger areas contain larger populations that are less likely to become extinct and large areas are more likely to become colonised by an organism moving randomly through the landscape.

A positive species-area relationship is common among terrestrial landscapes around the world (Ambuel and Temple, 1983; Howe, 1984; Vickery et al., 1994) and within Australia (Howe, 1984; MacNally and Watson, 1997; Seddon et al., 2001). In some cases this relationship has been confounded by the fact that more sampling effort has been concentrated in larger patches (e.g., Ambuel and Temple, 1983; Howe, 1984; Vickery et al., 1994). Our study and others controlled for this bias by using a fixed sampling effort in each patch regardless of area (MacNally and Watson 1997; Watson et al. 2000; Seddon et al., 2001; Major et al., 2001), and by sampling only one vegetative community (MacNally and Watson, 1997; Watson et al., 2000). When sampling effort is controlled for, species-area relationships tend to be less obvious (MacNally and Watson, 1997; Watson et al., 2000; Seddon et al., 2001), and this was the case in the present study.

In the present study, DECREASER and WOODLAND species richness was significantly positively related to patch area (Table 17; Fig. 35; Fig. 36). Generally, species most sensitive to habitat fragmentation in agricultural regions of southern Australia are insectivores, and some nectarivores (Reid, 1999; MacNally et al., 2000; Major et al., 2001; Seddon et al., 2001). Area-sensitive species that had an apparent area threshold of 10 ha in the present study were largely insectivorous species and were included in the DECREASER or WOODLAND species groups (Table 17). In contrast, generalist species located in remnants <10 ha were a mixture of habitat generalists and raptors and consisted of WOODLAND, OTHER and INCREASER species groups (Table 17). Species found in all or three of the four size classes were also mainly habitat generalists (Appendix 4).

Our finding that some insectivorous species were apparently area-sensitive could be explained by fragmentation effects, which are known to influence important life history attributes. For example, nest success and annual productivity of the rufous treecreeper (Climacteris rufa) were found to be significantly lower in a fragmented landscape when compared to continuous landscape (Luck, 2003). Although not statistically significant, there was an overall trend towards a lower juvenile survival rate in the fragmented landscape (Luck, 2003). The fragmented landscape was also found to have significantly lower prey biomass and provisioning rates to young (Luck, 2003). Zanette et al., (2000) found that in smaller patches female eastern yellow robins (Eopsaltria australis) received 40 % less food from males while incubating. This resulted in females leaving the nest to forage more regularly, and nestlings were provisioned with less food in smaller fragments. Other proximate factors known to influence bird species richness include time since isolation (Miller and Harris, 1977; Miller, 1978; Loyn, 1987; Soule et al., 1988), and the presence of large trees, which offer more diverse habitats, more food, and more hollows (Bennett et al., 1994; Gibbons et al., 2000; Ford et al., 2001; Seddon et al., 2001).

Fragmented landscapes containing many small, degraded patches may potentially act as population sinks (Donovan et al., 1995a, b; Major et al., 1999a; Zanette et al., 2000; Luck, 2003). Within these landscapes, there may simply not be enough source habitats to 'charge' habitat sinks (Major et al., 2001). A combination of a 'shortage of birds' as a result of a shortage of source habitat within fragmented landscapes, (Major et al., 2001) and deceased dispersal capabilities within these landscapes (e.g. Dunning et al., 1995; Belisle et al., 2001; Brooker and Brooker, 2002; Cooper and Walters, 2002) is thought to contribute to the depauperate nature of some species in fragmented landscapes. The extent of avifaunal declines in Australia may not be fully recognised, as some of these sink populations may become extinct due to the population within that patch reaching a new equilibrium after fragmentation (Bennett and Ford, 1997; Recher, 1999; Major et al., 2001). These further extinctions may take even longer to eventuate in Australia, as Australian birds are long-lived compared to birds of Europe and North America (Ford et al., 2001).

Patch area and abundance

Overall bird abundance was not related to patch area (Table 18), but INCREASER abundance was inversely related to patch area (Table 18; Fig. 37). A prediction of the equilibrium theory of island biogeography states that the abundance of an organism should be independent of patch size (MacArthur and Wilson, 1967). However, reviews of this phenomenon have yielded equivocal results (Bowers and Matter, 1997; Bender et al., 1998; Connor et al., 2000; Debinski and Holt, 2000). In Australia, some studies have reported an inverse relationship between bird abundance and patch area (Seddon et al., 2001), while others have reported a positive relationship between the two (Connor et al., 2000). An inverse relationship has sometimes been attributed to the effective habitat area of the species sometimes associated with smaller remnants (Haila, 1988; Laurance, 1991; Seddon et al., 2001). Abundance in habitat generalists (INCREASERS) was generally higher in smaller patches (Table 18; Fig. 37) (Traill and Duncan, 2000; Seddon et al., 2001). These species often feed outside the habitat patch, thus increasing their effective feeding area and giving a false impression of their abundance within small patches (Haila, 1988; Laurance, 1991; Seddon et al., 2001).

Edge Effects

Luck et al., (1999) distinguished three different types of reactions a species can have to habitat edges: (1) open-country species that generally inhabit open country in cleared habitat and only penetrate core habitat slightly; (2) edge-users whose abundance increases at the habitat edge; and (3) edge-avoiders whose abundance increases as the distance from the habitat edge increased towards the core habitat (Luck et al., 1999). One granivore and four nectarivores have previously been classified as edge-users, and one nectarivore and four insectivores were classified as edge-avoiders (Luck et al., 1999). This is consistent with the groups of species that have been found to be the most sensitive to habitat fragmentation throughout southern Australia (Reid, 1999; MacNally et al., 2000; Ford et al., 2001; Major et al., 2001; Seddon et al., 2001). In the present study, area to perimeter ratio, or the amount of 'core' habitat per unit edge, was significantly positively related to DECREASER and WOODLAND species richness (Table 19; Fig. 39; Fig. 40). However, we could not determine the influence of edge effects (alone) because total patch area and core habitat area were highly correlated, and thus confounded.

Habitat Complexity

Habitat complexity can potentially confound the effects of species-area relationships (Seddon et al., 2001) by positively influencing species prevalence, especially for WOODLAND or DECREASER species (habitat diversity hypothesis) (MacNally and Watson, 1997; Freudenberger, 1999; 2001; Seddon et al., 2001). In the present study, the lack of a significant relationship between patch area and habitat complexity, combined with the significant relationship of habitat complexity with WOODLAND species richness and abundance (Table 20; Fig. 43; Fig. 44), suggests that WOODLAND species richness and abundance were directly influenced by the complexity of the habitat, independent of patch area. Sites within the CIA that had a relatively extensive native understorey (e.g. Fig. 7; Fig. 8) were either not grazed by sheep or cattle, or were exposed only to relatively light grazing pressures (M. Robb pers. comm., Coleambally Irrigation Corporation). Barrett et al., (1994) considered patch condition to be "at least as important" as patch area in respect to the contribution to species richness in sites >6 ha. The fact that habitat condition has been found to contribute to species richness in this, and other studies has important conservation implications. For example, improving the habitat complexity or condition increase species richness without increasing patch size.

Landscape Variables and Birds

Once the potentially confounding effect of patch area had been included in the model exploring the effect of isolation on species richness, patch area explained the majority of variation in the data for WOODLAND and DECREASER species. Neither area nor any of the isolation measures contributed a significant amount to the variation in OVERALL or INCREASER species richness. Other studies examining similar habitat and landscape variables, and in similar degraded landscapes to this study, have found that habitat variables are generally more important than landscape variables in explaining the variation in species richness (Barrett et al., 1994; Watson et al., 2000; Major et al., 2001). Seddon et al., (2001) found that different components of a bird assemblage react to landscape variables in different ways. INCREASER species, as was found in the current study, had no distinct relationship between species richness and landscape attributes (Seddon et al., 2001). Species that exhibited a stronger relationship with landscape variables consisted largely of DECREASER and WOODLAND species (Seddon et al., 2001).

At a larger landscape scale, similar trends were evident in bird species reactions to landscape variables on the plains of northern Victoria (Bennett and Ford, 1997). In landscapes of mean size 277 km2, total tree cover was found to significantly influence WOODLAND species richness but not OTHER species richness (Bennett and Ford, 1997). Despite being highly intercorrelated, the relative effects of total tree cover on WOODLAND species were greater than that of measures of fragmentation (Bennett and Ford, 1997). The intercorrelation between these variables could partly explain why the modelled effects of the amount of tree cover in a landscape on the species richness of woodland species becomes accelerated after further reductions in tree cover at 10 % of tree cover remaining in the landscape (Bennett and Ford, 1997). Saunders and Ingram (1987) found that breeding success in the Carnaby's cockatoo (Calyptorhynchus funereus latirostris) suddenly dropped when <10 % of the surrounding landscape was cleared of heath habitat. Andren (1994) demonstrated through modelling random habitat clearance, that in landscapes with below 10 % tree cover, isolation between patches exponentially increases. As a result of this accelerating relationship in landscapes that are already severely fragmented and contain <10 % tree cover, further habitat loss will result in disproportionately high consequences as a result of this clearance. Thus, revegetation work undertaken in a highly degraded landscape would have a proportionally greater effect than that undertaken in a landscape with more habitat remaining. Fragmentation effects (reduction of patch area and isolation between patches) start to confound the effects of pure habitat loss at a level of around 10-30 % of the original habitat remaining in the landscape (Andren, 1994). At levels above this, species loss is said to be an effect of random sampling (Connor and McCoy, 1979; Andren, 1994; Ford et al., 2001). However, habitat is not always cleared at random and certain habitat types are lost disproportionately within a landscape, thus affecting species depending on that habitat in a disproportionate manner (Ford et al., 2001).

Connectivity in highly-degraded landscapes is important for dispersal and movement (Saunders et al., 1991; Bennett and Ford, 1997; Ford et al., 2001). Thirty km to the north of the Coleambally Irrigation Area (CIA) is the Murrumbidgee River, and 30 km to the south is Yanco Creek. Both of these waterways are associated with an extensive riparian corridor of vegetation. Bennett and Ford (1997) found that landscapes with low tree cover (<6%) on the northern plains of VIC, that maintained a rich assemblage of woodland birds, were usually those associated with the riparian zone of a major river or an area with an extensive network of minor creeks and roadside remnants. Extensive riparian corridors can function as biogeographic corridors (Bennett and Ford, 1997) and offer abundant and species rich bird assemblages (Fisher and Goldney, 1997; Oliver et al., 1999; MacNally et al., 2000; Ford et al., 2001). The corridors running to the north and south of the CIA could offer potential source populations for the CIA and surrounding landscapes and aid in dispersal and movement for migratory species. In a landscape roughly 100 km south of were the current study was undertaken Bennett and Ford (1997) also found a negative association between WOODLAND bird species richness and the prevalence of irrigation channels within the landscape. Bennett and Ford (1997) attributed this to the intensive nature of farming in these areas, small property size and the low amount of vegetation cover in these regions, as is found in the CIA, which only retains 2.5% vegetation cover.

The diversity of relationships to landscape and habitat variables in the current and other studies highlights the fact that different species react to the fragmentation of habitat in fundamentally different ways (Abensperg-Traun et al., 1996; Bennett and Ford, 1997; Ford et al., 2001; Major et al., 2001; Seddon et al., 2001). This diversity of reactions indicates the inappropriate nature of a strict adherence to the island biogeographic view in terrestrial matrix landscapes in most cases. Only in the minority of cases are remnants completely surrounded by an environment that is hostile to most of its inhabitants (McIntyre and Barrett, 1992). Environments can also vary in their hostility, as certain organisms can move quite freely through an environment that could be perceived as hostile to another (Ford et al., 2001). The largely inappropriate nature of island biogeographic theory to some species has led to the variegated landscape model (McIntyre and Barrett, 1992). These types of landscapes do not act as distinct ‘units’ to some species, and dispersal through such landscapes is not a matter of ‘life or death’ (McIntyre and Barrett, 1992; Ford et al., 2001).

The management of a variegated landscape would not involve managing the species assemblage as a whole, but to manage it in groups of species based on their reaction to fragmentation (McIntyre and Barrett, 1992). This is a difficult task, as the same species can react to fragmentation in a variety of ways across different landscapes across its distribution (Barrett et al., 1994; Bennett and Ford, 1997; Freudenberger, 1999; 2001; MacNally et al., 2000; Watson et al., 2000; Major et al., 2001; Seddon et al., 2001). The difference in a species’ reaction to fragmentation across landscapes can also occur over a relatively small scale of approximately 50 km (Freudenberger, 1999; 2001).

Relative Importance of Habitat and Landscape Variables to Birds

The relative influences of habitat and landscape variables on bird species richness in the present study seemed to indicate that habitat variables were more important in influencing the species assemblage at a given habitat patch than the spatial arrangement of patches in the near vicinity. Many studies have indicated that habitat variables are seemingly more important than landscape variables in influencing the species richness and prevalence within fragmented landscapes (Watson et al., 2000; Major et al., 2001; Seddon et al., 2001; MacNally and Horrocks, 2002). Other studies have noted that both habitat and landscape variables were important in influencing species richness and prevalence and also indicated that the relative effects of these variables can be highly species-specific (McGarigal and McComb, 1995; Knick and Rotenberry, 1995; Graham and Blake, 2001).

In the present study, one landscape variable and three habitat variables were found to contribute to the overall model that was developed relating to OVERALL species richness (Table 21). No landscape or habitat variables were found to influence OVERALL bird abundance. The cover of shrubs (0.5-2 m) was found to have a positive influence on species richness. Similar findings have resulted from other studies where the condition and structural complexity of the understorey influenced species richness, especially of some species types (Freudenberger, 1999; 2001: Seddon et al., 2001).

The current study indicated that the number of patches >10 ha within 2 km of the study patch, the number of dead trees and the number of regenerating overstorey trees all had a negative influence on bird species richness (Table 21). If the patches > 10 ha within 2 km of the study patch were of poor quality, they could have perhaps increased the dispersal capacities of invasive competitive species such as the noisy miner (Manorina melanocephala), thus producing a negative influence on bird species richness (Major et al., 2001). High numbers of dead trees were indicative of the advanced stages of dieback in the present study. Advanced stages of eucalypt dieback have been shown to strongly influence the abundance and species richness of birds (Ford and Bell, 1982). The negative influence of eucalypt dieback on species richness and abundance could also be compounded by the increased prevalence of noisy miners in this type of modified habitat (Ford and Bell, 1982; Loyn, 1987; Major et al., 2001; Seddon et al., 2001).

When the relative influences of landscape variables and patch area were examined, patch area was found to contribute significantly to the significant models that were created for DECREASER and WOODLAND species richness. This result is somewhat contrasted by the previous result where a landscape variable was found to contribute significantly when patch area did not (Table 21). This occurred because the relationship between patch area and species richness was of a log-linear nature, hence a linear regression conducted on untransformed data was not considered to be significant (Table 21). This study employed an area (log) versus species richness relationship, as have other studies (MacNally and Watson, 1997; Watson et al., 2000; Seddon et al., 2001). Thus, the area (log) and species richness (log) multiple regression will be more readily interpreted in the context of the other results of this study.

*Noisy Miner (*Manorina melanocephala*) Density and the Effect on Other Bird Species*

In contrast to other studies, noisy miner (M. melanocephala) abundance was not related to patch area (Table 22). This trend is unusual as many studies demonstrate a negative trend in Noisy Miner abundance with patch area (Loyn, 1987; Grey et al., 1997; 1998; MacNally et al., 2000; Ford et al., 2001; Major et al., 2001; Seddon et al., 2001; MacNally and Horrocks, 2002). Noisy Miner abundance also had no relationship with habitat edge, or area to perimeter ratio (Table 22). The lack of a distinct negative trend was again unexpected, as Noisy Miners are known to be edge-inhabiting species (Loyn, 1987; Ford et al., 2001). Our contrasting finding may be explained by the poor quality of some of the large remnants. Patches of degraded nature, especially those lacking an understorey and suffering from eucalypt dieback, are the preferred habitat of this species (Ford and Bell, 1982; Loyn, 1987; Major et al., 2001; Seddon et al., 2001). Seddon et al. (2001) indicated that the abundance of Noisy Miners was at its lowest levels (roughly 1.3 birds/ha) in large (>100 ha) shrubby remnants. They found that medium (10-100 ha) shrubby remnants had roughly half the densities (roughly 3.1 birds/ ha) as medium and small (<10 ha) degraded remnants (roughly 6.1 and 5.5 birds/ ha) respectively. As in the present study, the noisy miner was also found to be ubiquitous across remnants in the central sheep-wheat belt (Major et al., 2001). Major et al. (2001) indicated that only in remnants >200 ha were noisy miners found to be less prevalent. Remnants of this size were lacking in the present study.

In the present study, noisy miner abundance was not related to patch area or shape. Therefore, patch area and shape positive influenced insectivorous, WOODLAND and DECLINER species, independent of noisy miner abundance. Noisy miners have been identified as being one of the most important factors in determining the species richness and abundance of birds in fragmented landscapes in southeastern Australia by acting as an aggressive competitor species (Ford et al. 2001). However, species distribution, patch area, vegetation type and the presence of noisy miner are all interrelated (Grey et al. 1997, 1998; Ford et al. 2001; Major et al. 2001; Seddon et al. 2001).

Direct comparisons between noisy miner abundance and the abundance of some insectivores and nectarivores have indicated a density-dependent relationship, with an increase in noisy miner abundance, causing a compensatory decrease in insectivore and nectarivore abundance (Grey et al., 1997; 1998; MacNally et al., 2000; Major et al., 2001; Seddon et al., 2001). Using the same species groups as used in the current study, Seddon et al. (2001) found that noisy miner abundance had a significant negative effect on OVERALL species richness, WOODLAND species richness and DECLINER species richness. However, the only relatively strong trend that noisy miner abundance exerted on a species group in the current study was with DECLINER abundance (Table 22). The effect of interspecific competition as a result of noisy miners has been found to be at its greatest levels in remnants of <20 ha (Loyn 1987; MacNally et al. 2000).

Conclusions

The CIA region still maintains a relatively diverse avifauna considering the degraded state of the natural vegetation in the region. However, some insectivorous species proved to be relatively confined in their occurrence, and some key declining species of conservation concern were not located during this study. Five species identified as declining by Reid (1999) in the sheep-wheat belt of NSW were recorded in the region. One species listed as being nationally vulnerable and another three being listed as nationally near threatened were also recorded.

The overall species-area relationship was similar to that observed in other studies where the potentially confounding effect of increased sampling effort in larger patches had been controlled for. DECREASER and WOODLAND species richness was positively related to patch area and patch shape. The abundance of DECREASERS was also positively related to patch shape, while INCREASER abundance inversely related to both patch area and shape. As identified in other studies, the insectivorous guild was found to be the group of species that suffered from the effects of fragmentation in the most sensitive manner. A significant area effect was observed in bird assemblage composition. Three main groups of bird assemblages were identified in this study: Group 1 consisted of the pseudocontrol site; Group 2 consisted of small (<2.5 ha), relatively degraded sites; while Group 3 consisted of the remaining sites. A habitat complexity score that was developed for this study was significantly positively related to WOODLAND species richness and abundance, independent of patch area.

Landscape characteristics were found to contribute less to overall species richness and abundance than did patch area. Other studies indicate that in similar degraded agricultural matrix landscapes, patch characteristics generally contribute more to species richness than do landscape variables. The many different reactions of species to landscape and habitat variables that were encountered in this and other studies, highlights the fact that different species react to the fragmentation of habitat in fundamentally different ways.

In contrast to other studies, noisy miner (Manorina melanocephala) abundance was not related to patch area and/or the amount of edge present within a patch (patch shape), or to species richness or abundance of other species groups. However, the inverse relationship between noisy miner abundance and DECREASERS abundance approached statistical significance.

IV. Long-necked Turtles and Rice Farms

Why Move into Seasonally-available Rice Bays?

Our hypothesis that long-necked turtles invade seasonally-available rice bays in rice agroecosystems was supported by both radio-telemetry and captures. In natural systems, C. longicollis are known to reside in permanent water bodies such as lakes, but to then move into seasonally-available habitats such as ephemeral swamps (Kennett and Georges, 1990). In the present study, C. longicollis resided primarily in semi-permanent irrigation channels. However, soon after rice bays were irrigated, many turtles moved into them. Turtles returned to the channels after draining of the rice bays several months later. Thus, C. longicollis within rice agroecosystems displays a similar pattern of habitat use to that observed in natural systems.

In natural systems, an explanation for C. longicollis moving into ephemeral habitats was relatively clear: turtles in the permanent lakes experienced little or no growth or reproduction, apparently due to high densities relative to food availability (Kennett and Georges, 1990). From this example we hypothesized that C. longicollis moved into rice bays because invertebrate prey were more abundant and/or diverse in the bays than in the channels. However, this prediction was not upheld. Our invertebrate sampling indicated that rice bays did not offer a richer food source in terms of overall abundance and diversity of potential prey (Fig. 63). Rather, rice bays had significantly lower prey family diversity than the irrigation channels, which formed the core habitat of C. longicollis (Fig. 58). Thus, although the turtles may have moved to ephemeral rice bays to exploit higher invertebrate prey availability, this was not realised in terms of overall relative abundance or diversity of prey.

However, prey abundance/diversity per turtle, perhaps a more indicative measure of prey availability, may be higher for turtles once some individuals have moved into the rice bays. The additional food resources provided by the rice bays may be important in supporting current population sizes and for enabling growth and reproduction, by simply creating more feeding habitat. Large numbers of turtles remaining in the supply channel over the active season may lead to higher resource pressure and a lower net energy intake per individual than could be obtained if individuals also used rice bays. Therefore, the movement of C. longicollis into rice bays and other habitats over spring and summer may be important in obtaining sufficient food during a period when feeding rates are highest. Food shortage in over-wintering sites has been observed to induce annual movements to summer habitats in other species of turtles (Sexton, 1959; McAuliffe, 1978; Pluto and Bellis, 1988; Chase et al., 1989; Graham, 1995).

Alternatively, the movement of turtles into rice bays may reflect an evolutionarily - conserved behaviour in this species. The propensity of C. longicollis to move into ephemeral water bodies likely evolved in natural systems, where such movements are generally rewarded with a relatively high abundance of prey (Brinson et al., 1981; Kennett, 1987; Kennett and Georges, 1990). Because we were unable to demonstrate such a reward for C. longicollis in rice bays, it is possible that these movements reflect an innate propensity to exploit ephemeral water bodies which, although highly advantageous in natural systems, may not be beneficial in highly managed agroecosystems such as rice farms. We should note that an assumption of both hypotheses is that our prey availability sampling was representative. Further work comparing the prey availability between rice bays and irrigation channels would be useful, as prey abundance in rice bays has been found to vary depending on the type of water used, the source of the water, the water depth, biogeography, and the level of pesticides or herbicides used (Fasola and Ruiz, 1996). A study of the field energetics between turtles using the rice bays and those using irrigation channels would be particularly useful in determining the ultimate prey contribution of rice bays to C. longicollis.

Prey Availability: Catholic but with Preferences

In accordance with previous studies (Parmenter, 1976; Chessman, 1984b; Georges et al., 1986; but see Meathrel et al., 2002), our data showed that C. longicollis is a generalistic and opportunistic predator, consuming a wide range of prey types, including mostly invertebrates but also fish and frogs. However, C. longicollis in the present study became more selective of prey type in response to an increase in prey diversity (Fig. 64). The diet of turtles caught in the rice bays in November corresponded well with the prey availability in the both the rice tofo and the rice field in terms of composition and relative abundance, and this was reflected by a significant relationship between the abundance of prey types in the diet and their abundance in both the rice tofo and rice field for that month. However, in the next two months (December and January) there was no significant correlation between abundance of prey types in the diet and their abundance in the rice habitats, despite a progressive increase in both prey abundance and diversity from November to January in rice bays (Fig. 71).

The shift in prey selectivity is particularly well-illustrated by the response of turtles to chironomid abundance during these months. Chironomids comprised 25-85% of the total abundance of prey in the rice bays from November to January (Fig. 64). However, November was the only month in which turtles consumed chironomids in similar proportions to their abundance in the rice bays (Fig. 64). In December and January no chironomids were found in the stomachs of turtles from the rice bays, even though chironomids still dominated the total abundance of prey (Fig. 64). Instead, during both December and January there was a pronounced dietary shift towards new prey groups, mainly odonate and dytiscid larvae. These two groups comprised only 3% of available prey during December-January, yet they comprised over 60-80% of the total prey items consumed by turtles during those months (Fig. 64).

The relatively high consumption of odonate and dytiscid larvae may be due to their high profitability relative to the energy expended by turtles to obtain them. Bell (1990) found that coleopterans (the order containing dytiscids) have high energy content relative to other orders. Furthermore, odonates and coleopterans are generally larger than chironomids (Hawking and Smith, 1997). Since insect mass is positively correlated with energy content (Csada et al., 1992), C. longicollis may obtain a greater net energy intake per unit effort by consuming larger prey (presuming catchability is equal). Differences in visibility may have also contributed to the observed prey selection. Dytiscid larvae, despite their low abundance, were large and conspicuous and thus may be easier prey to find or capture. In contrast, the burrowing habits of chironomids (Hawking and Smith, 1997) may make them less visible to turtles. Georges et al. (1986) found that chironomids were one of the most poorly represented prey types in the diet of C. longicollis in a natural wetland system.

The trend towards increasing selectivity as a response to an increase in prey availability is in accord with optimal foraging theory. This theory is based on the premise that by foraging in a way that maximises energy intake, animals will increase their evolutionary fitness (Pyke, 1983). Optimal foraging theory predicts that when prey density is low, predators should employ a generalist feeding strategy and take prey in proportion to their abundance, while predators should select more profitable prey types when prey density is higher (Todd et al., 1998), and some turtle species have displayed such selectivity (Clark and Gibbons, 1969: Parmenter, 1980; Bury, 1986; MacCulloch and Secoy, 1983). The only other study to examine prey selectivity in C. longicollis found the diet of this species to be relatively unselective in relatively unproductive dune lakes (Georges et al., 1986).

Influence of Sex on Movement and Habitat Use

In the present study, sex influenced the timing and frequency of movements of C. longicollis between habitats. The majority of inter-habitat movements by males were in September to November (Fig. 60), which corresponds to the mating season (Parmenter, 1976; Chessman, 1978). This is likely to be a result of attempts by males to increase their mating opportunities since males can enhance their reproductive success by copulating with more females (Morreale et al., 1984). In contrast, females made very few movements between habitats during the mating period (Fig. 60). It may be worth reiterating that, once inseminated, females do not benefit from further mate searching (Morreale et al., 1984).

Females displayed a peak of between-habitat movements during November and December (Fig. 60), corresponding to the nesting season (Parmenter, 1976; Chessman, 1978). Reproductive success in females depends primarily on the quality of the habitat in which they place their offspring (Morreale et al., 1984). Thus, the increased frequency of movements between habitats made by females in the nesting period probably resulted from searching for nest sites. Females also displayed equally high between-habitat movements during March (Fig. 60) but this is likely to be related to the drainage of the rice bays, which triggered movement back to the supply channel.

The movement patterns described above correspond to the ‘reproductive strategies’ hypothesis (Morreale et al., 1984), which predicts that due to differences in reproductive roles, males would move greater distances and move between habitats more frequently than females during the mating season (Morreale et al.,1984). In the nesting season, however, the movement of females between habitats and the distances travelled should be equal to, or greater than, males (Morreale et al., 1984). This pattern has been found in a number of freshwater turtles (Morreale et al., 1984; Parker, 1984; Shubauer et al., 1990; Lovich et al., 1992; Brown and Brooks, 1993; Duda et al., 1999). In contrast to the pattern of movements between habitat types for each sex discussed above, the mean daily distances travelled by each sex bore little correspondence to the predictions of the ‘reproductive strategies’ hypothesis (Morreale et al., 1984). There was no statistical difference between the distances travelled daily by males and females between months or reproductive seasons.

The Trigger for Movements Between Habitats

Although the present study did not intend to ascertain the trigger for movements in C. longicollis, our observations and data indicated that water levels may have an important influence on movement patterns and habitat use. In the present study, decreases in water depth of about 30% appeared to trigger movement by C. longicollis to another water body. Four of the six cases involving turtles moving out of farm channels into supply channels appeared to be triggered by a temporary reduction in water depth. Similarly, although not standard practice, the rice bays on one farm were partially drained in mid-December to facilitate crop growth. This drainage of approximately 15 cm appeared to trigger movements away from this habitat. Three radio-tagged turtles were in the rice bays prior to this drainage. However, by mid-January all three had retreated to the supply channel. In contrast, a male (No. 11, Fig. 53) that used the rice bays on the farm northwest of the main study site, did not retreat from the rice bays during this period, probably because these rice bays did not undergo the same drainage.

Decreases in water level have previously been observed to trigger movement to other water bodies by C. longicollis (Chessman, 1978), and numerous other species of turtles (Cagle, 1950; Moll and Legler, 1971; McAuliffe, 1978; Gibbons et al., 1983). In natural systems, reduction in water level acts as a cue that prompts C. longicollis to retreat from seasonal habitats to more stable and permanent water bodies (Chessman, 1978; Kennett and Georges, 1990). It is worth mentioning that a reduction in shallow water levels during the growing season can be associated with an increase in water temperature. Unfortunately, in the present study no water temperature data were obtained due to a technical failure in data loggers. Although C. longicollis appears to be relatively unselective of its ambient temperature (Parmenter, 1976), further work should attempt to disentangle temperature from water reduction when investigating the trigger for turtle movements.

Implications and Recommendations

I. Managing On-farm Vertebrate Wildlife

A goal of the present study was to hypothesize on what attributes of rice farms underpin vertberate biodiversity, in order that we might maintain current biodiversity levels by managing that biodiversity. That is, we aimed to provide farmers with a knowledge base and recommendations on how to conserve on-farm wildlife. Although wildlife use of farms has received very little attention, there are currently considerable efforts targetted at better understanding habitat conservation and integrating it with farm management (Lindenmeyer et al., 2003). A complete understanding of the complex structure, composition, and function of on-farm wildlife would take decades or perhaps longer, and so there is no absolute prescription for improving biodversity on a farm (Salt et al., 2004). However, general patterns emerge after shorter periods of study (e.g., present study), and these patterns are often corroborated by research elsewhere. These corroborated patterns allow us to take a conservative approach to wildlife conservation through on-farm actions. Whenever possible, these actions should be conducted as experiments, whereby we learn of their effectiveness and apply an adaptive management approach.

Concurrent to the present study, a Biodiversity Strategy and Plan was developed for the Australian Rice Industry (Freudenberger and Stol, 2002). This plan included a number of practical recommendations for improving biodiversity on rice farms, based on current knowledge in biodiversity conservation in agroecosystems and elsewhere. Against this backdrop, we have highlighted six of these recommendations which were supported by our study, based on evidence of attributes underpinning the vertebrate biodiversity we sampled:

- in the garden, plant up a larger area and use a wide range of local shrubs, and mulch
- keep livestock out of woodland areas at the critical times when trees, shrubs, and grasses are germinating and becoming established
- use controlled grazing for grasslands and shrublands; don’t graze continuously
- revegetate in wide bands of trees (at least 5 rows wide), linking existing patches of vegetation.
- support or initiate improvements to vegetation beside roadsides or stock routes
- use channel banks for revegetation and providing habitat

Our findings support these six recommendations made by Freudenberger and Stol (2002), which can be lumped into broad management categories: (1) improving existing vegetation; and (2) revegetation.

(1) Improving Existing Vegetation

Evidence

Example 1: In our ‘birds and remnants‘ study, habitat complexity (e.g., mainly the presence and condition of understory shrubs) significantly influenced woodland bird richness and abundance, with more individuals and species detected in patches with a more complex understory.
Example 2: We noted that many of the Black Box depressions in both the extensive and ‘birds and remnants‘ study exhibited no regeneration. This is likely caused by grazing pressure and/or changes in hydrology. Grazing pressure can substantially reduce tree regeneration in Australian pastoral landscapes (Saunders et al., 2003), and has been shown to have negative effects on small ground-dwelling vertebrates in Australia and other parts of the world (reviewed in Fischer et al., 2004).
Example 3: We noted that on some farms, understory shrubs were limited to house gardens, and these gardens were the only area of the farm supporting species that require such structure (e.g., superb fairy wrens).

Example 4: Large numbers of birds, snakes, and frogs utilised vegetation along irrigation channels. In many cases this vegetation could be improved, and more heavily vegetated areas along channels harboured more of these animals.

(2) Revegetation

Evidence

Example 1: In our extensive study, we found that farms with vegetation remnant patches contained, on average: more species of vertebrates, reptiles, lizards, frogs, and birds (particularly woodland bird species). Lizards were also more abundant on farms with vegetation remnants.
Example 2: Large numbers of birds, snakes, and frogs, utilised vegetation along irrigation channels, yet many channels possessed no significant vegetation along their edges.
Example 3: In the ‘birds and remnants‘ study, we found that larger remnant patch area was associated with increased species richness of woodland birds, and declining birds.

What sized remnants are best for woodland and declining birds? Generally, our ‘birds and remnants ‘ study indicated that ‘bigger is better’. Larger patches of remnant vegetation held more species of woodland and DECREASER birds (Fig. 38). Close examination, however, revealed that while woodland bird species richness increases proportionally with an increase in patch area, the relationship between DECREASER bird species richness and patch area is curvilinear (Fig. 38). DECREASER bird species richness was no different between patches of < 2 ha and patches 2-10 ha in area (Fig. 38). In fact, our results indicate that protecting most DECREASER birds would require maintaining patches > 50 hectares (Fig. 38).

These results indicate that different strategies should be adopted, depending on which groups of birds are being catered for. If increasing the overall bird species richness is the goal, then any patch is better than none, and the bigger the patch the better, regardless of patch size. However, catering for DECREASER bird species would likely require creating vegetation patches > 50 ha.

In addition to these six recommendations (amalgamated into improving existing vegetation and revegetation), we also suggest the following recommendations, based on our studies.

(3) Conserve Existing Vegetation Remnants

We are unaware of the industry’s plans to expand rice production, but if the area of production expands, it should not be done so at the expense of native vegetation remnants, for the same reasons as (1) and (2) above.

(4) Maintaining Biodiversity in Rice Bays

Water extraction associated with irrigating rice and other crops has resulted in degradation of river health in the Riverina Bioregion (Norris et al., 2001). However, once this water is on farm, it becomes a ‘soup’ of bacteria and invertebrates, leading to a productive (albeit simplified) ecosystem with frogs, snakes, turtles, and aquatic birds that would not be in such high abundance without large amounts of on-farm water. *We are not asserting that the biodiversity values of this water on-farm offset the degradation to river health due to water extraction (see Norris et al., 2001 for a review of river health issues).* Rather, we are noting that once allocated and distributed this water is a valuable resource to on-farm vertebrate (and indeed, invertebrate) biodiversity. That biodiversity should be maintained, by considering impacts of herbicides, pesticides, and other farm practices. This also has implications for potential plans (and research) into cold tolerant varieties of rice that require less water. A wholesale change to a drier-based rice crop would be expected to have some impact on biodiversity of animals using rice bays on farms. Changes in irrigation practices impacted populations of frogs and waterbirds in Japan (Fujioka and Lane, 1997; Lane and Fujioka, 1998).

Evidence

Example 1: Richness and abundance of birds (mainly aquatic or semi-aquatic species) increased markedly with flooding of rice bays, with a similar decrease after draining.

Example 2: Long-necked turtles apparently use rice farms as a surrogate for natural wetlands, moving from the more permanent irrigation channels to the seasonally-available rice bays soon after flooding. A similar pattern of habitat use exists for the southern bell frog, Litoria raniformis, a threatened species inhabiting rice farms (Australian Museum Business Service, 2000; Pyke, 2002; S. Wassens, Charles Sturt University, unpubl .data.).

Example 3: Frogs were 1.5-2 times more abundant along rice bays than along dry crops (canola, fallow).

Example 4: The annual production of spotted grass frogs emerging from rice bays in the Riverina was estimated to be in the order of 500 million.

We would also like to note that the reason that our study does not support some of the other recommendations made by Freudenberger and Stol (2002) is because those recommendations are beyond the scope of our data. None of our findings are at odds with their recommendations, and many of their recommendations that we did not address in the present study are likely to be important.

For example, there is evidence that ground cover such as rocks, logs, bark, and leaves are important for small vertebrates (Goldingay et al., 1996; Lindenmeyer et al., 2003). Maintaining the presence of such cover should be beneficial to on-farm biodiversity. Conversely, removing fallen timber and rocks could adversely affect biodiversity of small vertebrates (Fischer et al., 2004). In another example, paddock trees have been shown to add significant value to biodiversity (Fischer and Lindenmeyer, 2002a; 2002b), and thus, whenever possible should be maintained and enhanced. Finally, protection, revegetation, and improvement of riparian vegetation along watercourses is likely to provide an essential corridor for wildlife (Lindenmeyer and Peakall, 2000).

II. The Importance of a Threatened Species Component of a Biodiversity Plan and Strategy for the Rice Industry

A central goal of the Australian rice industry’s environmental policy with regards to biodiversity is to ‘prevent further losses of biodiversity in the rice agroecosystem’ (Mullins and Sutherland, 2000); or to ‘conserve the variety of life within the rice-growing region of Australia’ (Freudenberger and Stol, 2002). This goal mirrors environmental policy with respect to biodiversity at the national, state, and local levels (e.g., Commonwealth of Australia, 1996; NSWNPWS, 1999), and is also consistent with Triple Bottom Line reporting (Commonwealth of Australia, 2003). Herein, we argue that this goal can only be achieved by recognising and managing species whose persistence are tenuous (e.g., threatened species).

Diversity has two readily measurable components, richness and abundance, and there is no argument that an ideal biodiverse landscape would consist of high numbers of many species. However, should these two components be treated equally with regards to managing biodiversity? If not, which is more important, or which deserves more attention if we are to maximise conservation efforts? These questions are important because they underpin any philosophy behind the allocation of resources toward biodiversity strategies and plans.

One way to amalgamate these two questions is to ask, ‘which would be more environmentally costly, a reduction in abundance or a reduction in richness? A direct comparison here is difficult, because the currency of abundance is the individual, and the currency of richness is the species.

However, genetic diversity among populations of a species is also important for its survival (Young and Clarke, 2000), and so extinctions at the population level threaten species richness, and so we could argue that the population could also be considered a currency of richness. We could never hope to agree on how many individuals of species A would be equivalent to an entire population of species B. Even if we could, it is difficult to envision a practical application of such a model, especially when considering whole assemblages of species in an ecosystem and associated problems quantifying their comparative 'worth'. However, we can compare their worth in a more qualitative sense.

The consequences of the loss of a species are irreversible. Barring some cloning technology, the species is lost forever, diminishing the biodiversity of a region by one unit, and resulting in a net loss to our rich environmental heritage. There may be further consequences in other species that interacted directly or indirectly with that species, and there may even be a negative economic consequence.

The loss of a population can also be irreversible, but not necessarily so. However, the challenges of restoring a population can be immense, and can compound the costs of future biodiversity conservation.

It is true that a reduction in abundance of a species can be a precursor to a loss of the population, and can therefore contribute to a loss in richness. However, in most cases species and populations can undergo a considerable reduction in abundance prior to there being any threat to their existence as a population or species. Therefore, for the sake or argument we consider those species here.

Suppose that we consider an assemblage of species living in a particular ecosystem. Like most ecosystems, there are past and current human impacts, resulting in (1) threatened species, (2) a reduction in habitat, influencing abundance in many species, and (3) a recognised need for biodiversity management. Given that the interested stakeholders possess limited resources, how should those resources be allocated to maximise biodiversity conservation?

Alternate strategies, simplified for the sake of argument, might include a threatened species plan, and a habitat restoration plan. The threatened species plan might include habitat improvements and restoration as well, but would also involve species or population monitoring, and any number of interventions (e.g., feral animal control, captive breeding) to ensure the species' survival. The habitat restoration plan might include revegetating or improving farms or riparian areas, or establishing corridors linking those areas. Both plans are critical for the maintenance of biodiversity in most human-affected ecosystems. Both plans are likely to result in an increase in abundance. However, restoration plans seldom are associated with a baseline inventory of species prior to intervention (i.e., revegetation) or a monitoring program, which together can assess changes in richness and abundance of species, thereby evaluating the success of the restoration effort. In some cases there is no assessment of impact or damage to ecosystem in the first place, casting doubt on the value of the restoration, relative to other opportunities in biodiversity conservation.

There are two ways to alleviate this problem associated with biodiversity strategies/plans that hinge on restoration efforts. The first way would be to invariably include an inventory and monitoring plan (I&M) in any restoration or revegetation effort. We recommend an I&M Program whenever resources allow it. The second way is to ensure that the plan includes a significant threatened species component. That is, a provision for research, management, and protection of species or populations in need of conservation.

Building a biodiversity that is (at least partly) centred around threatened species is the best way to reduce the probability of no net loss of species, or no net loss of biodiversity. Such a plan would identify the 'damage' and threatening processes involved from the species' point of view, allowing a targeted management strategy for the species involved.

This strategy is likely to be superior to that of a restoration plan alone, because the latter is based on our perception of what biodiversity requires, rather than an on-the-ground research. In other words, while restoration is generally positive with respect to biodiversity, without a supplemental strategy for the protection of species in need, it is likely to (1) fail to identify the needs of those species; (2) fail to monitor the assemblage or species in need; and therefore; (3) fail to protect threatened species, and; (4) leave us unsure of the value of our intervention.

We conclude that any biodiversity plan or strategy should explicitly include a threatened species component. Most or all biodiversity plans and strategies implicitly recognise threatened species by adhering to some central goal such as ‘the maintenance of current levels of biodiversity’, yet many do not cater for threatened species. Failing to prevent extinctions, either at the species or population level, would undermine any biodiversity plan or strategy. Promoting general biodiversity by making common or abundant species more common or abundant is desirable, but not at the expense of neglecting the species that are in decline or in need of conservation.

The Riverina Rice Farm Example

Much of the woodlands in the Riverina Bioregion have been cleared for human use, resulting in a paucity of isolated remnant vegetation (Benson, 1999; Eardley, 1999). Accordingly, species restricted to woodlands in and around the region are often rare or threatened, while species inhabiting more open habitats are generally common. The open habitat species are also usually more widespread and common across larger landscapes (e.g., across western New South Wales), and require less attention for their persistence. Many species inhabiting woodlands, on the other hand, are in need of management or protection, and their loss would equate to a net loss of the biodiversity in the region.

Because the majority of threatened species occupying rice farms (and generally) in the Riverina are likely to be woodland species, there is not likely to be many cases of threatened species with competing requirements on a particular rice farm. Improving or increasing the habitat of a threatened species is not likely to be detrimental to species occupying open habitats because those species are generally very common or abundant. Thus, managing threatened species on rice farms in the Riverina Bioregion could be considered a ‘safe’ conservation strategy, even if currently unproven for a given species.

Who is Responsible?

Managing threatened species should not be the responsibility of any one industry or stakeholder, but rather should be a joint venture encompassing as many community groups as possible. Further, there are too many threatened species for any one group to manage (for example, there are 69 threatened vertebrate species in the Riverina Bioregion; Ayers, 1995; EPBC Act, 1999; Mullins and Sutherland, 2001). The optimal strategy to conserve threatened species would be to formulate a regional threatened species plan, with a diversity of stakeholders including government, industry, and the general community (such a plan does not exist for the Riverina Bioregion or for most regions). In the absence of any formal regional threatened species plan, the next best approach would be for each stakeholder to invest in the conservation of a few threatened species. This strategy would (1) demonstrate a ‘duty of care’ of the stakeholder for biodiversity, and thus for environment; (2) encourage other stakeholders to follow suit, thereby promoting regional biodiversity conservation; and (3) raise the level of awareness of threatened species and threatening processes in the general community. In these ways the Australian rice industry would benefit greatly by initiating such a program.

III. Management Implications of the Birds and Remnants Study

Studies within Australia and abroad indicate that large patches and patches with good quality habitat are those that provide the best habitat for the conservation of bird species diversity (Major et al., 2001). Although the OVERALL species-area relationship has not proven to be as distinct in this and other studies (e.g. MacNally and Watson, 1997; Watson et al., 2000; Major et al., 2001), different components of the bird assemblage react to fragmentation and, specifically patch area, in fundamentally different ways (Ford et al., 2001; Seddon et al., 2001; MacNally and Horrocks, 2002). Within southern Australia a distinct group of insectivorous species generally are the most sensitive to fragmentation (Reid, 1999; Ford et al., 2001; Major et al., 2001; Seddon et al., 2001). Managing a landscape that would be beneficial to these species would require the retention or creation of large good-quality habitat patches (Freudenberger, 1999; 2001; Reid, 1999; Ford et al., 2001; Major et al., 2001; Seddon et al., 2001). Major et al. (2001) suggested the retention of patches >200 ha, and of good quality for the conservation of the insectivorous species group, although such retention is difficult on private land (Barrett et al., 1994; Seddon et al., 2001). Barrett et al. (1994) assigned conservation priority to patches of >20 ha. Other studies have suggested that priority should be given to patches of >10 ha with >20% shrub cover (Freudenberger, 1999; Seddon et al., 2001) and >10 ha with a “structurally complex” understorey (Freudenberger, 2001). Habitat “quality” has been indicated to be at least as important as the size of a patch > 6 ha for conserving species richness (Barrett et al. 1994). It is important to note that relatively “degraded” patches, even to the scale of paddock trees, offer important habitat for many species (Law et al. 1999; Gibbons and Boak 2000; Law et al. 2000).

Landscape characteristics did not prove to be as important as patch characteristics in the determination of bird species richness and abundance in this, and other studies (Watson et al. 2000; Major et al. 2001). The lack of large, good quality patches that could serve as population sources within the CIA is thought to be part of the reason for this finding. Most of the larger patches within the CIA have quite a low area to perimeter ratio and are of poor habitat quality (D. Bourne pers. obs.). Bennett and Ford (1997) demonstrated the value and importance of a good local network of riparian and/ or roadside vegetation and the proximity of a landscape to a major river riparian area to the species richness of WOODLAND birds.

The following management guidelines (Barrett et al., 1994; Freudenberger, 1999, 2001) are considered appropriate for the conservation of bird species richness and abundance within the agricultural context of the CIA, based on our results:

1) retain and maintain good quality remnant patches > 10 ha (however, patches > 50 ha are best, and any patch is better than none
2) if revegetation work is to be undertaken: (a) maximise the vegetation complexity by planting overstorey and understorey species; (b) maximise the core area of plantings; and (c) plant in conjunction with existing patches if possible
3) connect the landscape along natural river corridors by linking existing patches and minimising grazing pressure within these regions
4) exclude stock from existing state forests and crown land to encourage regeneration and minimise soil compaction
5) minimise the removal of fallen timber and dead trees from habitat patches.

IV. Management Implications of the Turtles in Rice Study

Although C. longicollis is both widespread and common in the Riverina Bioregion (e.g., Chessman, 1984b; Chessman, 1988; Meathreal et al., 2002), its commonness on rice farms is unknown, and management practices could be beneficial in maintaining or boosting turtle populations in rice agroecosystems. The present study has established that within rice agroecosystems the ability of C. longicollis to range over large areas and move between rice bays, farm channels, supply channels and farm dams, is an important aspect of their ecology in these landscapes. According to Parmenter (1976), land managers can facilitate overland travel in C. longicollis in two ways. First, because the use of wire netting fences appear to be the major factor in preventing turtles from reaching water and safety when on land, this fence type could be discontinued. Secondly, because ground cover may provide important refuges for turtles in overland travel, the provision of shrubs or bushes on otherwise bare expanses of grazed land by landholders may potentially be advantageous to turtle populations in the area.

A specific feature of rice farms that may pose a threat to C. longicollis is the dethridge wheel. This is a wheel with rotating blades which gauges the amount of water provisioned to farms from supply irrigation channels. In the present study we observed notable injuries in some turtles including four turtles with cracked shells, one turtle missing a foot and two turtles with several missing toes. A few farmers and irrigation operators have indicated that these wheels are responsible for frequent injury and mortality in the turtles (e.g., David Marsden, rice grower; Bob Patton, Murray Irrigation Limited, John Ginnazin, Goulburn-Murray Water, pers. comm.). Turtles can be crushed by the spates of the wheel as it rotates. Turtles killed in this manner will often block the dethridge wheels, causing logistical problems for irrigation management (Bob Patton, Murray Irrigation Limited, personal comment). The extent to which this occurs needs to be investigated and quantified. If the results indicate that the mortality rate is high enough to warrant action, an option may be to modify the structure by slotting a steel grid in front of the gate. This will not restrict the flow of water but will prevent entry by certain sizes of turtles (the sizes that the grid precludes would depend on the size of the gaps in the grid). Such a structure may well provide benefits to both turtle populations and irrigation management.

In summary, because rice is one of the world’s most extensive and important crops (Chang and Luh 1991), understanding how animals use rice farms is critical to maintaining biodiversity in agroecosystems. The results of the present study reinforce the potential for rice agroecosystems to provide valuable habitat for wildlife. Specifically, our results have demonstrated that C. longicollis uses all major aquatic habitats of rice farms. Within the rice agroecosystems C. longicollis displays similar movement patterns and habitat use to those in natural systems, residing primarily in irrigation channels but readily moving into seasonally-available rice bays. Unexpectedly, rice bays did not provide greater relative prey abundance or diversity than the channels. Further work is needed to determine if this movement pattern is beneficial to the turtles through an increase in prey availability per individual turtle, or simply artifactual, conserved behaviour that evolved in natural systems prior to European settlement in this long-lived species.

Appendices

Appendix 1

Recordings of Australian frog calls

Australian Frog Calls: Subtropical East
Produced by John N. Hutchinson.
(http://www.naturesound.com.au)

Australian Frog Calls: Tropical North-east
Produced by David Stewart -- Nature Sound.
(http://www.naturesound.com.au)

Australian Frog Calls Volume 1
Produced by David Stewart -- Nature Sound.
(http://www.naturesound.com.au)

Frog Calls of North-eastern New South Wales
Produced by David Stewart -- Nature Sound.
(http://www.naturesound.com.au)

Frog Calls of the Greater Sydney Basin
Produced by David Stewart -- Nature Sound.
(http://www.naturesound.com.au)

Appendix 2

Questionnaire for farmers

Biodiversity on Rice Farms Survey

Contact Details (optional): Name_______________Ph.__________

Significant Species

Have you seen any of the species below on your property in the last 5 years? (please circle)

Narrow-nosed Planigale **Y/N**

Southern Bell Frog **Y/N**

Inland Carpet Python **Y/N**

Gould's Sand Goanna **Y/N**
(note yellow tail tip)

Habitats for wildlife

Do you have any remnant patches of woodland or shrubland on your property? ______________________________

Is your property along a wooded creek or river?_____________

Location

What is the nearest town to your property?________________

Appendix 3

Species located during surveys and their associated species code and species recorded incidentally in the region (in bold). National status follows EPBC listings, status follows Reid (1999).

Common name	Scientific name	National status	Status
Emu	*Dromaius novaehollandiae*		Decreaser
Stubble Quail	*Coturnix pectoralis*		
Australian Wood Duck	*Chenonetta jubata*		
Pacific Black Duck	*Anas superciliosa*		
Australian Grebe	*Tachybaptus novaehollandiae*		
Little Pied Cormorant	*Phalacorcorax melanoleucos*		
Little Black Cormorant	*Phalacorcorax sulcirostris*		
Great Cormorant	*Phalacorcorax carbo*		
Australian Pelican	*Pelecanus conspicillatus*		
White-faced Heron	*Egretta novaehollandiae*		
Little Egret	*Egretta garzetta*		
White-necked Heron	*Ardea pacifica*		
Great Egret	*Ardea alba*		
Intermediate Egret	*Ardea intermedia*		
Nankeen Night Heron	*Nycticorax caledonicus*		
Glossy Ibis	*Plegadis falcinellus*		
Australian White Ibis	*Threskiornis molucca*		
Straw-necked Ibis	*Threskiornis spinicollis*		
Royal Spoonbill	*Platalea regia*		
Yellow-billed Spoonbill	*Platalea flavipes*		
Black-shouldered Kite	*Elanus axillaris*		
Square-tailed Kite	*Lophoictinia isura*		
Black Kite	*Milvus migrans*		
Whistling Kite	*Haliastur sphenurus*		
Spotted Harrier	*Circus assimilis*		
Swamp Harrier	*Circus approximans*		
Brown Goshawk	*Accipiter fasciatus*		
Collared Sparrowhawk	*Accipiter cirrhocephalus*		
Wedge-tailed Eagle	*Aquila audax*		
Brown Falcon	*Falco berigora*		
Australian Hobby	*Falco longipennis*		
Grey Falcon	*Falco hypoleucos*	Near Threatened	
Black Falcon	*Falco subniger*		
Peregrine Falcon	*Falco peregrinus*		
Nankeen Kestral	*Falco cenchroides*		Increaser
Black-tailed Native-hen	*Gallinula ventralis*		
Black-winged Stilt	*Himantopus himantopus*		
Masked Lapwing	*Vanellus miles*		

Whiskered Tern	*Chlidonias hybridus*		
Rock Dove	*Columba livia*		
Common Bronzewing	*Phaps chalcoptera*		
Crested Pigeon	*Ocyphaps lophotes*		Increaser
Peaceful Dove	*Geopelia striata*		
Galah	*Eolophus roseicapilla*		Increaser
Little Corella	*Cacatua sanguinea*		
Sulphur-crested Cockatoo	*Cacatua galerita*		
Cockatiel	*Nymphicaus hollandicus*		Increaser
Superb Parrot	*Polytelis swainsonii*	Vulnerable	
Crimson Rosella	*Platycercus elegans*		
Eastern Rosella	*Platycercus eximius*		Increaser
Australian Ringneck	*Barnardius zonarius*		
Blue Bonnet	*Northiella heamatogaster*		Increaser
Red-rumpped Parrot	*Psephotus haematonotus*		Increaser
Budgerigar	*Melopsittacus undulatus*		
Pallid Cuckoo	*Cuculus pallidus*		
Fan-tailed Cuckoo	*Cacomantis flabelliformis*		
Horfield's Bronze-cuckoo	*Chrysococcyx basalis*		
Barn Owl	*Tyto alba*		
Australian Owlet-nightjar	*Aegotheles cristatus*		
Laughing Kookaburra	*Dacelo novaeguineae*		
Red-backed Kingfisher	*Todiramphus pyrrhopygia*		
Sacred Kingfisher	*Todiramphus sanctus*		
Rainbow Bee-eater	*Merops ornatus*		
Dollarbird	*Eurystomus orientalis*		
Supurb Fairy-wren	*Malurus cyaneus*		
Striated Pardalote	*Paralotus striatus*		
Weebill	*Smicrornis brevirostris*		
Western Gerygone	*Gerygone fusca*		
Inland Thornbill	*Acanthiza apicalis*		
Chestnut-rumped Thornbill	*Acanthiza uropygialis*		Decliner
Buff-rumped Thornbill	*Acanthiza reguloides*		
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*		Increaser
Yellow Thornbill	*Acanthiza nana*		
Spiny-cheeked Honeyeater	*Acanthagenys rufogularis*		
Striped Honeyeater	*Plectorhyncha lanceolata*		
Little Friarbird	*Philemon citreogularis*		
Blue-faced Honeyeater	*Entomyzon cyanotis*		
Noisy Miner	*Manorina melanocephala*		Increaser
Yellow-throated Miner	*Manorina flavigula*		Increaser
Singing Honeyeater	*Lichenostomus virescens*		
White-plumed Honeyeater	*Lichenostomus penicillatus*		Increaser
Painted Honeyeater	*Grantiella picta*	Near Threatened	
Black Honeyeater	*Certhionyz niger*		
Jacky Winter	*Microeca leucophaea*		Decliner
Red-capped Robin	*Petroica goodenovii*		Decliner
Grey-crowned Babbler	*Potmatostomus tempporalis*	Near Threatened	Decliner
Rufous Wistler	*Pachycephala rufiventris*		Decliner
Restless Flycatcher	*Myiagra inquieta*		Decliner

Magpie-lark	*Grallina cyanoleuca*	Increaser
Grey Fantail	*Rhipidura fuliginosa*	
Willie Wagtail	*Rhipidura leucophrys*	Increaser
Black-faced Cuckoo-shrike	*Coracina navaehollandiae*	Increaser
White-winged Triller	*Lalage sueurii*	
Masked Woodswallow	*Artamus personatus*	
White-browed Woodswallow	*Artamus superciliosus*	Decreaser
Dusky Woodswallow	*Artamus cyanopterus*	
Grey Butcherbird	*Cracticus torquatus*	
Pied Butcherbird	*Cracticus nigrogularis*	Increaser
Australian Magpie	*Gymnorhina tibicen*	Increaser
Australian Raven	*Corvus coronoides*	Increaser
Little Raven	*Corvus mellori*	Increaser
White-winged Chough	*Corcorax melanorhamphos*	Increaser
Apostlebird	*Struthidea cinerea*	
Singing Bushlark	*Mirafra javanica*	
Richard's Pipit	*Anthus novaeseelandiae*	
House Sparrow	*Passer domesticus*	Increaser
Zebra Finch	*Taeniopygia guttata*	Increaser
Mistletoebird	*Dicaeum hirundinaceum*	
Welcome Swallow	*Hirundo neoxena*	Increaser
Tree Martin	*Hirundo nigricans*	
Clamorous Reed-warbler	*Acrocephalus stentoreus*	
Rufous Songlark	*Cinclorhamphus mathewsi*	
Brown Songlark	*Cinclorhamphus cruralis*	
Golden-headed Cisticola	*Cisticola juncidis*	
Common Starling	*Sturnus vulgaris*	Increaser

Appendix 4

Species recorded during surveys and their percent occupancy across size classes and across all sites (overall).

Species	<2 ha (n=4)	2-10 ha (n=7)	10-50 ha (n=7)	>50 ha (n=5)	Overall
Apostlebird	0	29	14	40	22
Australian Hobby	0	0	0	40	9
Australian Magpie	100	72	100	100	91
Australian Owlet-nightjar	0	0	0	20	4
Australian Raven	100	86	100	100	96
Australian Ringneck	25	14	0	20	13
Black Honeyeater	0	0	0	20	4
Black Kite	25	0	14	0	9
Black-faced Cuckoo-shrike	75	43	72	100	70
Black-shouldered Kite	25	29	0	0	13
Black-tailed Native-hen	75	14	14	20	26
Blue Bonnet	25	86	57	60	61
Brown Falcon	25	14	29	20	22
Brown Goshawk	25	0	0	0	4
Budgerigar	25	0	0	0	4
Buff-rumped Thornbill	0	0	14	0	4
Chestnut-rumped Thornbill	0	0	14	60	17
Clamorous Reed-warbler	25	0	14	20	13
Cockatiel	75	86	72	60	74
Collared Sparrowhawk	25	0	0	0	4
Common Bronzewing	0	14	14	0	9
Common Starling	100	100	100	60	91
Crested Pigeon	100	100	100	80	96
Crimson Rosella	25	0	0	0	4
Eastern Rosella	0	43	29	60	35
Emu	0	0	0	40	9
Galah	75	100	86	100	91
Grey Butcherbird	25	43	72	100	61
Grey Fantail	0	0	0	60	13
Grey-crowned Babbler	0	29	43	60	35
Horfield's Bronze-cuckoo	0	0	0	20	4
House Sparrow	75	29	14	0	26
Inland Thornbill	0	0	0	20	4
Jacky Winter	0	0	0	20	4
Laughing Kookaburra	25	14	43	0	22
Little Friarbird	0	0	0	20	4
Little Raven	25	43	29	20	30

Magpie-lark	100	100	100	100	100
Masked Woodswallow	25	14	0	20	13
Mistletoebird	0	0	29	20	13
Nankeen Kestral	25	29	14	20	22
Nankeen Night Heron	0	14	14	20	13
Noisy Miner	50	100	100	100	91
Pacific Black Duck	25	14	57	20	30
Pallid Cuckoo	0	0	14	0	4
Pied Butcherbird	0	72	72	20	48
Rainbow Bee-eater	0	0	0	20	4
Red-capped Robin	0	0	0	20	4
Red-rumpped Parrot	100	100	72	80	87
Restless Flycatcher	0	0	14	20	9
Rufous Wistler	25	0	14	40	17
Sacred Kingfisher	25	14	29	20	22
Singing Honeyeater	0	14	0	0	4
Spiny-cheeked Honeyeater	0	14	0	20	9
Striated Pardalote	50	86	100	100	87
Striped Honeyeater	0	43	29	20	26
Superb Parrot	0	43	57	40	39
Supurb Fairy-wren	0	29	14	20	17
Weebill	0	29	57	80	43
Welcome Swallow	25	0	0	0	4
Western Gerygone	0	0	0	40	9
White-browed Woodswallow	25	14	0	20	13
White-faced Heron	50	57	57	20	48
White-necked Heron	25	0	0	0	4
White-plumed Honeyeater	75	72	29	20	48
White-winged Chough	0	43	43	60	39
White-winged Triller	0	14	14	20	13
Willie Wagtail	75	72	72	60	70
Yellow Thornbill	25	14	57	60	39
Yellow-rumped Thornbill	25	0	43	60	30
Yellow-throated Miner	0	14	0	0	4
Zebra Finch	25	14	14	20	17

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