

ECOLOGY AND CONSERVATION OF THE PURPLE-CROWNED FAIRY-WREN (*Malurus  
coronatus coronatus*) IN THE NORTHERN TERRITORY, AUSTRALIA

By

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This work is dedicated to my son Lex

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By

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The purple-crowned fairy-wren (*Malurus coronatus coronatus*) is a small endemic habitat specialist restricted to riparian vegetation in northern Australia. Its range extends from the Kimberley in Western Australia east to the Victoria River District in the Northern Territory. One of two subspecies, *M. c. coronatus*, is a threatened species with a fragmented distribution attributed to habitat loss and degradation.

Along the Victoria River, the purple-crowned fairy-wren inhabits stands of canegrass or river grass (*Chionachne cyathopoda*). The results from my study show that, in the Victoria River District, the purple-crowned fairy-wren is highly dependent on *Chionachne* for cover, foraging and breeding and was not found in any other dominant vegetation. This habitat association is in contrast to the single previous study, where the purple-crowned fairy-wren was primarily dependent on the aquatic pandanus (*Pandanus aquaticus*).

Purple-crowed fairy-wrens live in small family groups and groups often remained in the same territory for consecutive years. Territories are best described as polygons with an average size of 0.41ha. The average group size was 2.6, and although groups comprising a single breeding male and female were the most common (50%), groups with a single additional helper

were also common (41%). Purple-crowned fairy-wrens often foraged in loose family groups, usually within 2 meters of the ground, and favored river grass and freshwater mangrove (*Barringtonia acutangula*) as foraging substrates.

My study found purple-crowned fairy-wrens had a long breeding season of 22 weeks, the start of which coincided with the end of the wet season with a peak in May. During my study, 65 nests were found. The majority of nests (85%) were located in river grass with an average nest entrance height of 40.1cm. The average clutch size was 2.9 with 88% of clutches consisting of three eggs. Incubation was estimated at approximately 14 days and the nestling period 10 days. A high nest predation rate of at least 51% was found during my study, with most predation events occurring at the egg stage. Overall, the mean number of fledglings produced by a breeding female was 1.07 per year.

One of the primary disturbances investigated during my study was cattle grazing. Adult survivorship was heavily influenced by intense grazing resulting in a 74% loss of adult birds within one year in addition to a significant reduction in group size. Such a high mortality rate is very alarming for a species where the annual adult survival rates can be as high as 90%, with an average of 61%. Adult mortality is likely attributed to a loss of cover as mean river grass height was greatly reduced and the percentage of bare ground increased at grazed sites.

.A helicopter survey (encompassing 170 km of river frontage) along the Victoria River revealed that the primary habitat of the purple-crowned fairy-wren is extremely fragmented and patch size extremely variable ranging from 0.23 hectares to 244 hectares with an average of 24.5 hectares. Distance between patches ranged from 0.3 km to 41 km with an average distance of 1.37 kilometers.

The population of the purple-crowned fairy-wren in the Victoria River District is characterized by a fragmented distribution and high sensitivity to intense grazing pressure. High nest predation rates found during identify the need further investigation to determine primary nest predators. My study was conducted during high rainfall years thus constituting a “best case scenario.” The results from my study indicate that conservation efforts need to focus on reducing grazing pressure and increasing habitat connectivity to ensure the long-term survival of this species in the Northern Territory.

## CHAPTER 1 GENERAL INTRODUCTION

This chapter provides a brief introduction to the biology of the species forming the focus of my study. This chapter also introduces, justifies and contextualizes the primary research questions addressed in the remainder of the thesis.

### **Background**

The Maluridae is a family of birds endemic to Australia and New Guinea; the family includes five genera, three of which are found in Australia (Rowley and Russell, 1997). The fairy-wrens (*Malurus*) have been an overall well-studied genus in Australia, and have provided a rich and fertile research area for Australian ornithology, leading to much of the information on cooperative breeding, longevity, and brood parasitism (Schodde, 1982; Tidemann, 1986; Rowley *et al.*, 1991; Rowley and Russell, 1995; Brooker and Brooker, 1996; Rowley and Russell, 2002). For some of the southern *Malurus* species, banded populations have been maintained for more than 13 years providing invaluable insight into the adaptation of this family to the highly variable environmental conditions of the Australian continent, and exemplifying life history traits of endemic Australian passerines (Rowley *et al.*, 1991; Brooker and Brooker, 1994; Russell and Rowley, 1998).

The Australian continent has been subjected to severe landscape alteration since the arrival of first European settlers with the more populated, southern regions particularly affected (e.g. Ford *et al.*, 2001; Hobbs, 2005). In this context, fairy-wrens have been the subject of research examining the effects of habitat perturbations and fragmentation (Brooker and Brooker, 1994; Russell and Rowley, 1998). As small territorial understory species with long lifespans, fairy-wrens lend themselves well to any studies examining the effects of habitat alterations.

Despite the large number of studies on fairy-wrens, the research focus has been very uneven and there has been little previous attention given to the only threatened *Malurus* species. There are two recognized subspecies of the purple-crowned fairy-wren showing variation in plumage and size; the western (*M. c. coronatus*) and the Carpentarian (*M. c. macgillivrayi*). The two subspecies are separated by approximately 200 kilometers of unsuitable habitat (Figs. 1-1 and 1-2). My study does not directly consider the latter subspecies.

The western race of the purple-crowned fairy-wren (*M. c. coronatus*), hereafter PCFW, is currently listed as vulnerable by the Australian government (Environment Protection and Biodiversity Conservation Act, 1999). It has a patchy distribution in the Kimberley of Western Australia and the adjacent Victoria River District in the Northern Territory. The PCFW is a riparian habitat specialist that has experienced a substantial decline and is now exceedingly fragmented in distribution (Smith and Johnstone, 1977; Rowley, 1993; Garnett and Crowley, 2000). Due to its specific habitat requirements and threatened status, the PCFW is a prime candidate for studying the impacts of some of the primary threats to biodiversity in northern Australia, in particular in riparian zones. The status, ecology and conservation management of the PCFW are the subject of my study.

The PCFW has been the focus of only one previous detailed study (Rowley, 1993; Rowley and Russell, 1993). That study was based on the Drysdale River in Western Australia (Fig. 1-2) in riparian habitat dominated by aquatic pandanus (*Pandanus aquaticus*), a woody palm-like shrub that may form dense thickets, typically with foliage 2-5 m high, that grows in and immediately adjacent to watercourses (Fig.1-3). In that study, wrens were seldom encountered more than 10 meters away from the river. Thus far, knowledge of the PCFW in the Northern Territory is restricted to only a few preliminary surveys conducted along the Victoria River

(Boekel, 1979; Parks and Wildlife Commission of the Northern Territory). As a threatened species with a fragmented distribution this represents a serious gap in knowledge and one which my study aims to address.

In contrast to the habitat features and association evident in Rowley's study of the PCFW in the Kimberley, on the Victoria River and its tributaries, particularly in the lower reaches, the PCFW shows a strong affinity for habitat dominated by River Grass *Chionachne cyathopoda* (more commonly referred to as canegrass) [a tall (to 2-5m. robust grass forming almost monospecific and very dense stands: Fig. 1-3], and to a lesser extent northern canegrass *Mnesithea rottboellioides*. This apparent marked difference in habitat relationships affords a particular opportunity for comparative ecological study. Further, these apparently different requirements may necessitate site-specific management guidelines, rather than perhaps inappropriate extrapolations for a different environmental setting.

Purple-crowned fairy-wrens are territorial year-round residents, living in either pairs or small family groups often maintaining the same territory year after year (Rowley and Russell, 1993). Although socially monogamous, it is likely they are sexually promiscuous as are many other fairy-wrens although genetic data confirming this is still lacking. Like other *Malurus* species, they are cooperative breeders. Helpers are often male progeny that remain in the natal territory after reaching maturity, while females usually disperse before the start of the following breeding season (Rowley and Russell, 1993).

As a small endemic territorial species restricted to riparian habitats in northern Australia, the PCFW has without doubt been witness to many landscape changes since the first European settlers arrived in the 1880s (Rowley, 1993). As has been the case with many bird species in northern Australia, the potential response of the PCFW to disturbances is poorly understood, but

it is likely that the changes wrought by European colonists have been generally unfavorable to PCFWs (Thomas 1964; Rowley, 1993). This may be particularly so because the PCFW is a riparian specialist and European use of land has predominantly focused on rivers and riverine environments (Sattler, 1993).

Relatively little information exists about the historical distribution of the PCFW in the Northern Territory. This can mostly be attributed to the remoteness of its range, the relative recency of European settlement, and the sparse human population of the region. To this day much of its range is largely inaccessible due to the topography and lack of roads. Even along the Victoria River, only a few areas are reliably accessible by road, and even these may be unreachable in the monsoonal wet season. Access by boat is hindered by the numerous rock bars that are dispersed throughout the river system, and river access is also complicated by the abundance of large saltwater crocodiles (*Crocodylus porosus*).

The scarcity of accurate data on the historical distribution of the PCFW substantially constrains the assessment of population trends since settlement, and of current trends in the population. Although there are some anecdotal and observational data on the landscape changes that have occurred in riparian habitats since European settlement, until very recently there was little quantitative data available on these changes (Lewis, 2002; Start and Handasyde, 2002; Sharp and Whittaker, 2003; Sharp and Bowman, 2004).

In northern Australia, agriculture has been generally less intensive and the landscape remains more sparsely populated than in most other regions of Australia (Ridpath, 1985). However, pastoralism (livestock grazing) and changes to fire regimes have both been implicated in changes to faunal composition in northern Australia (Burbidge and McKenzie, 1989; Barnard, 1925; Franklin, 1998). Current research is revealing that extensive and often subtle changes in

the landscape have altered the distribution and abundance of many species and that these changes are extremely complex and often interrelated (Blakers *et al.* 1984; Franklin, 1999; Fraser, 2000; Williams *et al.*, 2002). Further, from studies of the role of multiple disturbances, and in particular exogenous ones (Dobkin *et al.*, 1998; Woinarski and Ash, 2002; Krueper *et al.*, 2003), it is clear that multiple and often interrelated disturbances are at the root of these changes in observed patterns, making it difficult to identify one single causal factor.

Riparian ecosystems encompass many diverse values including economic, social and biological (Gregory *et al.*, 1991; Malanson, 1993). Rivers and associated riparian ecosystems play an integral role in the maintenance of healthy regional ecosystems, and have been referred to as a “lifeline” or “aorta” (Wilson, 1979; Gregory *et al.*, 1991; Knopf and Samson, 1994). Riparian landscapes, as an ecotone between the terrestrial and aquatic area, represent a zone of high biodiversity where habitat alterations can have extensive impacts (Knopf *et al.*, 1988; Naiman *et al.*, 1993; Saab, 1999; Krueper *et al.*, 2003; Scott *et al.*, 2003).

Riparian zones provide habitat for both residential and migratory species and may provide a refuge or act as a corridor for dispersal for non-obligate riparian terrestrial species (Machtans *et al.*, 1996; Bentley and Catterall, 1997; Fisher and Goldney, 1997; Skagen *et al.*, 1998; Woinarski *et al.*, 2000). Riparian corridors can also be the linkage between multiple habitats in a mosaic of landscapes, enabling species to move through the landscape accessing suitable habitat (Machtans *et al.*, 1996; Skagen *et al.*, 1998). This is especially important in northern Australia with its extended dry season, where the riparian vegetation can be of great importance to a large number of species when suitable habitats may be isolated from one another. Furthermore, in northern Australia, riparian corridors may allow species to extend their range into lower rainfall areas (Woinarski *et al.*, 2000). For these reasons, it is not surprising that studies have shown that avian

diversity and abundances in riparian zones tend to be highly dynamic, particularly influenced by seasonal cycles (Knopf and Samson, 1994; Woinarski *et al.*, 2000).

Riparian habitats are exposed to both endogenous (natural) and exogenous (unnatural) disturbances at more intense rates than surrounding habitats (Planty-Tabacchi *et al.*, 1996). Endogenous disturbances, such as flooding and other changes in water flow, are routine events in most river systems in northern Australia. However, exogenous disturbances, such as weed invasions and grazing, have been experienced by these systems only fairly recently from an evolutionary perspective. In some instances disturbances can be considered both exogenous and endogenous depending on the circumstances. For example, fire caused by a lightning strike in the early wet season can be considered an endogenous disturbance, although it can be classified as an exogenous disturbance when it is set in the middle of the dry season to provide “green pick” for cattle.

One of the primary disturbances being investigated in northern Australia is fire and changes resulting from altered burning regimes (Braithwaite, 1987; Bowman, 1998; Yibarbuk *et al.*, 2001). Response to fire is dependent on fire regime (frequency, intensity, and timing) and ecological characteristics, thus research on multiple species and assemblages has demonstrated decidedly differently responses to fire (Anderson, 1991; Griffiths and Christian, 1996; Williams *et al.*, 2002). Altered burning regimes appear to be at the core of major changes in the avian assemblage in Australia (Garnett, 1992; Woinarski, 1990; Woinarski and Recher, 1997; Woinarski and Ash, 2002), especially among granivorous birds in northern Australia (Franklin, 1999; Fraser, 2000). For the endangered gouldian finch (*Erythrura gouldiae*), an increase in late season (i.e. hot) fires leading to a decline in suitable nesting trees was identified as one of the factors implicated in the widespread decline of this species in northern Australia (Tidemann,

1996). More recently, research has demonstrated that fire impacts gouldian finch populations by affecting seed availability in the short-term and altered floristic structure in the long-term (Price *et al.* in press).

Introduced species continue to be one of the primary threats to biodiversity throughout the world and Australia has not escaped this problem (Soulé, 1990; Vitousek *et al.*, 1996; Pimental *et al.*, 2000). In Australia, an average of 10 plant species become naturalized each year (Groves, 1997), many of which have been intentional introductions (Lonsdale, 1994). These introduced plants have invaded, and continue to invade, native landscapes where they can have devastating results and in extreme cases alter ecosystem function (Braithwaite *et al.*, 1989; Gordon, 1998). For example, in northern Australia, the introduced grass *Andropogon gayanus* has become a threat to biodiversity in the region by modifying vegetation composition due to the effect it has had on the grass-fire cycle (Rossiter *et al.*, 2003). Riparian zones in particular may be more susceptible to plant invasions than adjoining landscapes due to combination of factors that include ease of propagule spread, habitat continuity, disturbed soil, and high nutrient levels (Griffin *et al.*, 1989; Malanson, 1993; Planty-Tabacchi *et al.*, 1996).

In northern Australia, pastoralism may have had a profound influence on ecosystem composition, function and structure (Archer *et al.*, 1988; Whitehead, 2000; Sharp and Whittaker, 2003). Riparian systems may be particularly affected: pressure is greatest during the extended dry season when cattle and feral animals congregate along the riparian corridors (Myers *et al.*, 2001; Lewis, 2002). Grazing has been implicated in influencing avian composition in riparian zones, most frequently resulting in an overall reduction in species richness (Taylor, 1986; Saab *et al.*, 1995; Krueper *et al.*, 2003; Scott *et al.*, 2003). In addition, Ammon and Stacey (1997) recorded higher avian nest predation rates at grazed sites compared to ungrazed sites. Grazing

impacts may be particularly severe for avian species that occupy the lower levels of the vegetation rather than canopy dwelling species as structure of the understory can be greatly modified by grazing (Schultz and Leininger, 1990). For instance, foraging height was found to be a good indicator for determining avian species response to grazing, with the majority of ground-dwelling species showing a decline with increased grazing pressure (Martin and Possingham, 2005).

In many cases, it becomes hard to attribute major landscape changes to one single type of disturbance, and potential confounding effects make identifying causes even more difficult. Weed invasion, grazing and fire all seem to play a role in shaping the shifting vegetation composition within the riparian landscape. For example, the increase in woody vegetation in the riparian zone within the tidal portion of the Victoria River has been attributed to a lack of fire which has arisen as a consequence of extreme overgrazing (Sharp and Whittaker, 2003). In this instance both fire and grazing are at the crux of the observed change in vegetation composition that now seems to be irreversible.

My dissertation has three objectives. Firstly, as our ecological knowledge of the PCFW thus far has been primarily based on a single study, I collected ecological knowledge of the species in a contrasting setting that could then be used to fulfill the other objectives of my study.

The second objective was to examine the potential response of PCFW to grazing. Two other disturbances, weeds and fire, are probably also important but are dealt with more peripherally in this thesis.

The third objective was conservation-based and species-specific; to develop a suitable conservation management strategy that aims to ensure the long-term survival of the PCFW. Although my third objective was species-specific, by using a habitat specialist as an indicator

species, management recommendations will potentially benefit other species within the riparian habitat. A synthesis of the first two objectives was necessary to achieve this objective.

### **Study Aims and Layout**

Chapter 2 provides a background of the study region including historical, environmental, and social factors that have shaped the region. For the PCFW, an historical background and habitat associations in the study are discussed. A description of the study sites is also provided in this chapter.

The specific research questions addressed and approaches used are as follows:

- Is the ecology of PCFW in systems dominated by *Chionachne* (canegrass) different to that described in the only previous study (where the system was dominated by *Pandanus*)?

This question is addressed in Chapter 3.

Data on the general ecology of the PCFW were collected and compared to previous research based in a distinctly different habitat. These data also provide the foundation for the subsequent questions posed in this thesis.

- What is the status of *M. c. coronatus* within the study area?

This question is addressed in Chapter 5 with additional information from Chapter 3 (PCFW ecology) and Chapter 4 (disturbances).

Based on the data obtained from a helicopter survey, the distribution of *Chionachne* was mapped. Territory and group sizes of PCFW at specific study sites were used to assess PCFW habitat requirements and densities. Other suitable habitat areas, outside of the specific study sites, were surveyed to determine PCFW presence or absence. I used these data to estimate population size and to map the distribution of the PCFW in the study region.

- Is the current population of the purple-crowned fairy-wren stable?

This question is addressed in 5, but relies on results found in Chapter 3 and 4.

The density, reproductive success and adult survivorship of the PCFW, supplemented by *Chionachne* distribution, are the key indications used to determine the stability of the PCFW population.

- Do disturbances, in particular grazing, have an impact on the population of the purple-crowned fairy-wren?

This question is addressed in Chapter 4.

Grazing effects were analyzed using comparative means as grazing pressure was distinct between sites. Habitat variables were used for these comparative analyses. Reproductive success, adult survivorship and group size are the variables analyzed to detect any differences between sites. Because initial observations indicated that PCFW frequently used the introduced weed *Xanthium* as a foraging substrate, foraging observations were conducted to determine if this weed is a positive habitat component for the PCFW.

- What are the best management strategies for this species to ensure long-term survival?

This question is addressed in Chapter 5.

Management guidelines were developed based on (1) habitat requirements of the PCFW based on both the ecological and distribution data; (2) areas most suitable for rehabilitation and preservation with regard to connecting fragmented habitat based on the *Chionachne* distribution survey; and (3) recommendations for multiple stakeholder involvement based on personal observations, interactions that have occurred at field sites, and existing policies and practices.

### **Obstacles and Constraints**

This is a largely correlative study, with little opportunity to manipulate experimental and disturbance variables. It was impossible to manipulate most of the disturbances that occurred at

study sites. For example, my initial study design included a balanced representation of grazing intensity. However, without advance warning, one landholder introduced a high stocking rate of cattle to my main ungrazed sites during the course of my study. By the end of the three year study, there were only two sites left that were not subjected to grazing and even these had seen an increase in some disturbance agents (e.g. an increase in feral buffalo).

Resourcing constraints common to all PhD projects meant that my study was conducted over a three year period. This is a brief window through which to view the biology of a species likely to live more than six years; and in a region where irregular rainfall events may mean that there are no typical years or typical ecological responses.

Despite these constraints, the data I collected have some distinct advantages. The ‘realism’ of the data makes them more pertinent for devising a suitable management plan by giving an indication of important factors which may or may not be controllable by management. The implications of my research are that stakeholder attitudes toward protecting riparian zones will have a great influence over any future conservation efforts.

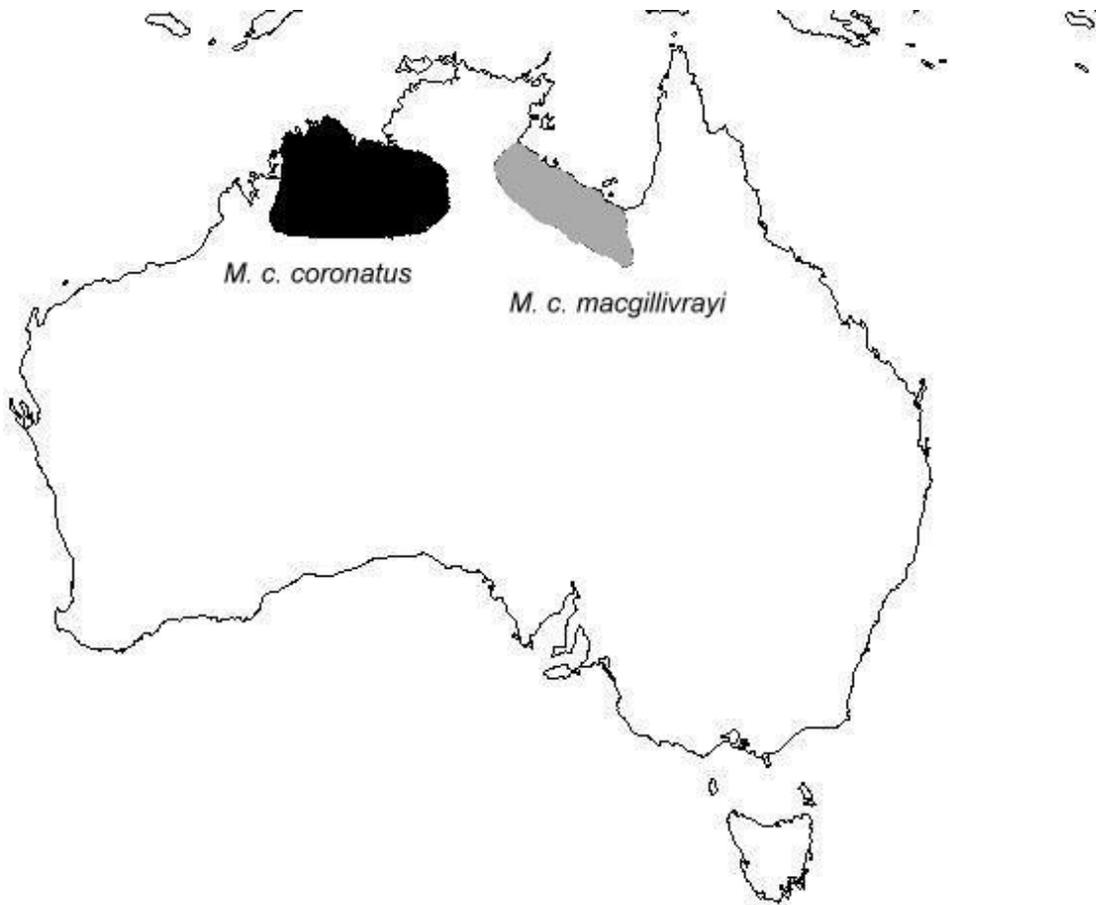


Figure 1-1. Distribution of the two subspecies of *Malurus coronatus* (adapted from Rowley and Russell, 1997). Note that the actual area of occupancy is far less than that indicated in this map, as the distribution is much more circumscribed to a network of narrow riparian strips (unmappable at this scale) rather than the broad polygons indicated here.

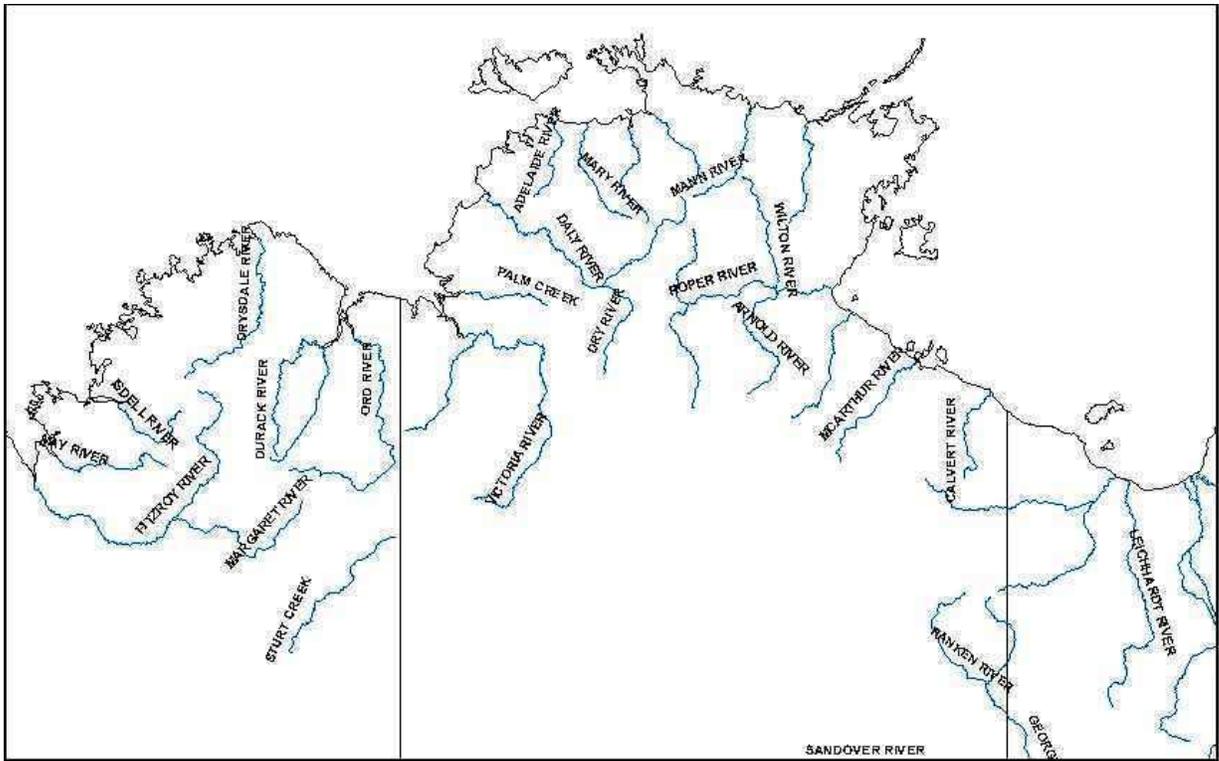


Figure 1-2. Major drainage lines within the distribution of the two species of the purple-crowned fairy-wren *Malurus coronatus*. The western subspecies (*M. c. coronatus*) has been found from the Fitzroy River in Western Australia east to the Victoria River in the Northern Territory. The Carpentarian subspecies (*M. c. macgillivrayi*) has been found from the Leichhart River in Queensland north to the Roper River in the Northern Territory.



Figure 1-3. Pandanus (*Pandanus aquaticus*).



Figure 1-4. Canegrass (*Chionachne cyathopoda*).

CHAPTER 2  
THE VICTORIA RIVER DISTRICT AND THE PURPLE-CROWNED FAIRY-WREN;  
NATURAL ENVIRONMENT AND HUMAN DIMENSIONS WITHIN THE STUDY AREA

This Chapter introduces the study area providing a description of location, climate, topography, geology and vegetation. From a human dimensions standpoint, it provides a description of the history, land use, and tenure of the region. In relation to the PCFW, the brief historical background of this species in the Victoria River District (hereafter VRD) is a reflection of the scarcity of records. The habitat description of the PCFW highlights the lack of any intensive surveys of this species in this region. This Chapter concludes with a description of the location and condition of the individual field sites used during my study. The environmental, social and historical aspects of the VRD and the PCFW described in this chapter form an important base for this thesis.

**The Victoria River District**

**Natural Environment**

The Victoria River District, in the semi-arid tropics of northern Australia, encompasses approximately 125,000 km<sup>2</sup> of land located 500 km south-west of Darwin with multiple land tenures in place including national park, pastoral, aboriginal, defense, and private lands (Hill, 2000; Karfs, 2000) (Fig. 2-1).

The Victoria River itself is approximately 720 km in length that originates on Riveren Station and eventually drains into the Joseph Bonaparte Gulf. The mouth of the river is more than 10 kilometers wide with an extensive tidal influence reaching more than 500 kilometers upstream, approximately 20 kilometers past Timber Creek (Kirby and Faulks, 2004). The Victoria River is a fairly shallow river with multiple sand and rock bars, which during the dry season separate the river into multiple billabongs, especially in the upper reaches. The river generally flows for only four to six months when flooding can be extensive. Annual flow

volume has been recorded as high as 4,184 million m<sup>3</sup> or a mean discharge of 909m<sup>3</sup> per second at the Victoria River bridge, adjacent to one of the study sites, GNP1 (Kirby and Faulks, 2004).

The average yearly rainfall for this region is approximately 800 mm (decreasing from the coastal north to the inland south: 938.5 mm in Timber Creek and 648.4 mm at Victoria River Downs) with a distinct wet and dry season (Bureau of Meteorology, Australia, 2005) (Figs. 2-2, 2-3). The wet season typically lasts from November through to March with the highest rainfall recorded in January and February when flooding most frequently occurs. Daytime temperatures average around 30<sup>0</sup> C in the dry season and 37<sup>0</sup>C in the build-up (the characteristically hot and humid time preceding the wet season) and wet season, although temperatures over 40<sup>0</sup>C are not uncommon in the build-up (Fig. 2-4, Bureau of Meteorology, Australia, 2005). Humidity levels vary according to the distinct seasons, ranging from a mean relative low of 40% in the dry season to a mean relative high of 88% in the wet.

There are five main catchments within the VRD, of which the Victoria River is the largest. The VRD encompasses characteristic mesas and sandstone plateaux, tropical savannas, and rocky outcrops with spinifex (*Triodia* sp.). About half of the region consists of very shallow soils (e.g. rugged stony outcrops, plateaux and hills), whilst the rest are erosional and alluvial plains that are favored for pastoralism (Kraatz, 2000). The shallow (or skeletal) soils belong to the Pinkerton land system whereas the alluvial floodplains belong to the Ivanhoe land system consisting of cracking clays (Stewart *et al.*, 1970). Major drainage lines, such as the Victoria River, are predominantly duplex clay-dominated soils. Table 2-1 illustrates the relationship between the parent material and the associated soils found in the region.

Multiple vegetation types characterize the VRD, and their distinction is associated with soils and annual rainfall (Fig. 2-5). Widespread vegetation types include *Eucalyptus* dominated

woodlands, mixed species low woodland, eucalyptus dominated low open woodland and grasslands, each with their own associated understory grasses. Common tree and grass species found in association with each broad vegetation type are listed in Table 2-2.

Less extensive than these vegetation types, there is a narrow network of riparian vegetation alongside the main river courses. This vegetation is described in more detail in the following sections.

### **Human Dimensions**

Aboriginal people have lived in the VRD for at least 40,000 years and there are multiple clans in the district, with approximately 24 language groups, that have a strong physical, spiritual, and social connection to the land (Riddett, 1990; Mulvaney and Kamminga, 1999). Approximately 20% of the land in the VRD is aboriginal-owned and there are about 27 communities in the region, although many of these consist of small family groups. There are also six aboriginal-owned pastoral properties that cover approximately 10% of the district (Davis, 2000).

The first European to officially explore the Victoria River was John Stokes who arrived aboard the HMS Beagle in 1839 led by Captain John Wickam. However, they did not stay in the area very long nor travel far upstream and their descriptions of the flora and fauna are fairly superficial (Lewis, 2002). From 1855 to 1856 the North Australian Expedition led by Augustus Gregory explored the VRD in much greater detail. Ferdinand von Mueller, a botanist, was a member of this expedition and his reports have valuable descriptions of the vegetation in the district (Mueller, 1858).

Pastoralism is one of the most important industries in the Northern Territory and there are approximately 1.4 million head of cattle in the territory with the majority (45%) destined for the live export trade to South East Asia (Peart, 2001). In the VRD, pastoralism is the major industry

and 59% of the land is used for grazing. There are over 20 pastoral properties ranging in size from 1,100 km<sup>2</sup> to 12,000 km<sup>2</sup>; and the total herd in the VRD is estimated to range from 515,000 to 750,000 head of cattle (Hill, 2000).

Undoubtedly, pastoralism has had a profound influence on the VRD ecological landscape. Management depended on an open or free-range system and overstocking was rampant with no established paddocks being present to control stock (Lewis, 2002). As a consequence of this primitive management practice, large numbers of cattle congregated along riparian corridors and billabongs causing great damage to these systems. According to Sharp and Bowman (2004), many of the changes that have occurred along the riparian alluvial plains along the Victoria River are irreversible.

The region has been heavily influenced by pastoralism since the arrival of the first European settlers in 1883. The initial relationship between the early settlers and the aboriginal people was a troubled one that included many violent altercations (e.g. Davis, 2000; Riddett, 1990). However, by the 1950s, the cattle stations depended greatly on the local aboriginal people as workers although they were only paid in rations and were in generally treated very poorly. The situation of virtually free aboriginal labor came to an end after the famous walk-off by aboriginal people at Wave Hill station in 1966. This was followed by similar strikes throughout the district in the 1970s that revolutionized aboriginal rights (Davis, 2000).

On Victoria River Downs Station alone, cattle numbers rose from 15,000 in 1889 and to between 119,000 to 170,000 in 1921. In addition to cattle, feral donkey and horse numbers also soared putting further pressure on the ecosystem. For example, on Victoria River Downs in 1960 there were an estimated 20,000 donkeys and in the 1940s there were an estimated 20,000 scrub bulls (Lewis, 2002). With no supplementary water available away from the river, the

pressure on the riparian zones during the extended dry season must have been tremendous. Conditions improved dramatically in the late 1970s and early 1980s with the commencement of the Brucellosis and Tuberculosis Eradication Campaign (BTEC) that resulted in improved fencing, control of feral cattle, and increasing provision of water away from riparian zones removing some of the pressure from these systems (Karfs and Trueman, 2005).

### **The Purple-Crowned Fairy-wren in the Victoria River District**

Most of the information that exists for the region is in some way related to pastoralism and there is a paucity of information on biodiversity. Hence, in relation to the PCFW there are few data on distribution or abundance until fairly recently. However, accounts given by some of the earlier explorers give an indication of the type of vegetation found along the Victoria River and its tributaries. These accounts also give an insight into the condition of the riparian areas previous to settlement and pastoralism, but only on a superficial level.

Photographs comparing various sites in the VRD published by Lewis (2002) also provide an insight into changes that have occurred in the landscape although these mainly demonstrate an increase in tree cover and any changes in riparian grass cover is not discernable. In addition, the majority of early pictures were taken when pastoralism was already well established and many of the sites had already been grazed so they give no indication of pre-European settlement condition (Lewis, 2002).

In relation to the PCFW, one of the most interesting observations that earlier explorers made is the abundance and density of “reeds” along the banks of rivers and creeks. These reeds were obviously common and widespread as they are described and referred to on many different occasions by the early explorers in the region including John Stokes in 1839 and Augustus Gregory in 1855 (but see Lewis 2002, for multiple excerpts). Lewis (2002) has inferred that these reeds are most likely *Chionachne cyathopoda* and the description of them being almost

impenetrable, is in my experience, an accurate assessment. If the reed is indeed *Chionachne*, then it points to a potentially widespread distribution of the PCFW in the VRD before the arrival of European settlers in the region (see below for a description of the primary habitat of the PCFW).

### **Primary habitat of the Purple-crowned Fairy-wren in the Victoria River District**

In the past, there has been a great deal of confusion in regard to the primary habitat of the PCFW, and this confusion extends to the VRD as well. In areas where the PCFW inhabits *Pandanus aquaticus* this has not been a problem but in the VRD where it inhabits riparian grasses multiple species have been erroneously listed as the primary habitat of the PCFW. The majority of the confusion appears to lie in the varied usage of common plant names by different authors over time. In particular, the common name canegrass has been used to describe a variety of riparian grass species in association with the PCFW.

In the VRD the primary habitat of the PCFW is the riparian species river grass or canegrass *Chionachne cyathopoda*, sometimes in conjunction with northern canegrass *Mnesithea rottbeolioides* (Table 2-3). *Chionachne cyathopoda* is a rhizomatous perennial species with a widespread tropical and subtropical distribution and often forms dense thickets along riparian corridors with average height of 2.7 meters (at ungrazed sites), although it can reach a height of up to 5 meters (Cowie *et. al.*, 2000). In the VRD, *Chionachne cyathopoda* is an important source of food for many bird species and large flocks of finches (e.g. star finch *Neochmia ruficauda*, crimson finch *Neochmia phaeton*, yellow-rumped manakin *Lonchura flaviprymna* and chestnut-breasted manakin *Lonchura castaneothorax*) congregate in the grass when it is seeding (pers. obs.)(see Appendix 1 for a complete list of bird species).

Along the Victoria River, dense stands dominated by *Chionachne* can extend more than a kilometer away from the river along drainage lines, but it is generally found within 200 meters of

the river bank. During flooding events, *Chionachne* stands may be completely inundated and flattened. However, it is generally resilient, and recovers rapidly following such flood events (pers. obs.). Areas that are inundated over long periods (e.g. drainage depressions) usually do not contain *Chionachne* suggesting it does not survive long periods of inundation (pers. obs.).

I did not encounter any areas where PCFW's were found in grass stands dominated by *Mnesithea* alone although these types of sites were surveyed. Several authors have listed *Mnesithea* as the primary habitat along the Victoria River; however this seems highly unlikely as we surveyed the same areas only to find *Chionachne* as the dominant grass species. I assume that this discrepancy is related to species identification and not a shift in habitat utilization.

Habitat that was dominated by *Pandanus aquaticus* was also surveyed (approximately 30 kilometers in various areas were surveyed including Sullivan's Creek, East Baines River and Wickam River), however, I did not locate PCFW's in this habitat despite this vegetation type being abundant. In the past the PCFW has been found in *Pandanus* habitat along the Wickam River (Boekel, 1979), I surveyed this area but was unsuccessful in locating any wrens in that area. It is possible that they no longer occur there or occur at very low densities. This is not particularly surprising as Boekel (1979) described the low density of birds and expressed concern for the future survival of that population. There are reports of scattered populations in *Pandanus* along the upper reaches of the Victoria River but as was reported by Boekel (1979) in low numbers.

The common name canegrass has been widely used to describe a range of riparian grass species in association with the PCFW throughout its range (McGill, 1970; Boekel, 1979; Rowley, 1993; Goodfellow, 2001). As is often the case with common names, they can represent a large number of species which may vary in structure, ecology and distribution. Species that

have been referred to as canegrass in association with the PCFW include *Ophiuros exaltatus* (similar to *Mnesithea rottboellioides*), native bamboo *Bambusa arnhemica* and exotic bamboo *Bambus* sp., northern canegrass *Mnesithea (Coelorhachis) rottboellioides*, and various *Sorghum* sp.

Within PCFW habitat, tree species consisted primarily of flooded box or coolibah *Eucalyptus microtheca*, river red gum *Eucalyptus camaldulensis*, with some leichardt pine *Nauclea orientalis* at the canopy level and freshwater mangrove *Barringtonia acutangula*, ficus *Ficus coronulata* and native white currant or dogwood *Flueggia virosa* at the middlestory level. Additional species found at lower densities at some sites included bauhinia *Bauhinia cunninghamii*, cathormion *Cathormion umbellatum*, gutta-percha tree *Excoecaria parvifolia*, weeping paperbark *Melalueca leucadendra*, and the aquatic pandanus *Pandanus aquaticus* (See Appendix 2 for a list of riparian plant species).

Other species very common among the stands of *Chionachne* and *Mnesithea* included a variety of introduced weed species. These included noogoora burr *Xanthium strumarium*, castor oil *Ricinus communis* and wild passionfruit *Passiflora foetida*. By far the most abundant weed and probably of greatest concern was noogoora burr. Introduced in the 1970s around Pigeon Hole (Lewis, 2002), *Xanthium* has rapidly become widespread throughout the Victoria River catchment area. This species and its potential impacts on the PCFW are discussed in greater detail in Chapters 4 and 5. Two other weed species, hyptis *Hyptis suaveolens* and rubber bush *Calotropis procera* are also found commonly in this habitat but are generally restricted to vegetation further away from the river and outside of *Chionachne* stands.

## Study Sites

### Selected Study Sites and Conditions

During the first (preliminary) field season in 2001, I investigated various locations of the Victoria River and tributaries to identify potential study sites that contained PCFW and were accessible most of the year. A considerable amount of time was spent in discussion with stakeholders to familiarize them with the goals of this research project and to gain property access. I surveyed locations that contained both *Chionachne* and *Pandanus* as all the previous research indicated *Pandanus* was the primary habitat. A total of 12 sites were evaluated as potential field sites, of these, four did not contain any PCFW's and access was denied to three other sites, resulting in five field sites that were usable (Fig. 2-6).

My aim was to establish both multiple un-grazed and grazed (preferably with differing weed densities) sites for comparisons. Although this was the experimental design preference, and initially sites were set up under those specifications, the following field season revealed that most of the sites no longer adhered to the presumed and/or original conditions, or sites could no longer be used. For example, after conversations with landholders several sites were selected on the understanding that they were fenced and hence ungrazed. However, subsequent visits revealed that most of the fences were not maintained and cattle were present. Due to the floods that occurred every year during my study, fences were continuously washed away and damaged, deeming them ineffective for excluding cattle from the riparian zone. Maintenance of fencing was an issue throughout the Victoria River catchment and many sites that I surveyed during the preliminary field season had damaged fencing.

During the first field season I chose five sites. One of these, a pastoral property with heavy grazing pressure that still retained PCFW groups, I identified as suitable and fieldwork had commenced at this site when the landholder decided against allowing continued access to the

property. In lieu of this site, in the second year of my study I identified another site that had also been heavily grazed, Dashwood Crossing on Victoria River Downs was used for the remainder of my study (Fig. 2-6). Unfortunately, this site was less suitable due to the distance from other sites, lack of access at the beginning of each season, and small number of wren groups in residence. Changes in grazing pressure, erosion, and weed infestation during the different field seasons are summarized in Table 2-4.

The first site (Coolibah) was located at the Coolibah Crocodile Farm (15.56256, 130.94884) on the eastern bank of the Victoria River downstream from Gregory National Park. This small parcel of land is privately owned and the riparian zone is only used for river access and as a low impact campsite during the dry season, although only a small clearing is used for camping. This site initially (2001) contained a very dense understory of *Chionachne* and PCFW's were easily heard and observed throughout the site and it was deemed a high quality habitat for PCFW with little erosion, low density of weeds and no known grazing pressure.

In July 2002, approximately 20 head of cattle from the neighboring station gained access to this site and it was grazed intensively, the cattle remained on the site until the beginning of the wet season (approximately October). In 2003, cattle once again gained access to this site and grazed the site but with less intensity. On both occasions, cattle grazed intensively on *Chionachne* and spent most of their time within the riparian zone where they had easy access to water. Once the *Chionachne* had been fed upon, less desirable plants (ones that are known to be toxic in large doses) were also eaten and included *Xanthium* and *Ricinus*.

My second site (Dashwood) was located at Dashwood Crossing on Victoria River Downs Station (16.33598, 131.11253), and was situated more than 200 km by road from the other sites and it was not possible to collect as much data from this site due to distance and limited access

during the wet season. *Chionachne* abundance at this site was low, especially in comparison to the other sites used in my study. *Chionachne* was found in small patches and sometimes in single clumps spread inconsistently through the riparian zone.

The Dashwood site had been consistently grazed for many decades, was heavily eroded and had high densities of weeds (in particular *Xanthium*). In recent times this site has been fenced and grazing pressure lowered but cattle still access the site at times, most likely due to damaged fences. According the current VRD manager and a long-term pastoral facilitator (D. Hill pers. comm.) erosion has subsided greatly since many areas have been fenced and new paddocks have been developed, resulting in reduced grazing pressure. The vegetation remained fairly constant at this site and no obvious changes took place year to year.

The third site (Fitzroy) was located on Fitzroy Station (15.56195, 130.92899) on the western bank of the river not far downstream from Coolibah Crocodile Farm and is an Aboriginal owned station and a small family group resides at the site. In 2001 there was a very dense understory of *Chionachne* that was almost impenetrable in many places. Although it had previously been used for grazing only a few head of cattle were present in 2001. PCFW were observed in large numbers and I deemed this site as high quality PCFW habitat. There was minimal activity in the riparian area with the owners of the property only going down to the river for fishing and to refuel the water pump located next to the river. There were a few weeds present on the side of the dirt track leading down to the river and in some of the flood gullies.

In 2002 this site was leased to a pastoralist and cattle grazed the site at light to moderate levels until the beginning of the wet season. In 2003, some cattle were still present at the site but in very low numbers since paddocks and bores had been established at sites away from the river reducing pressure in the riparian zone. However, in 2003 there was an influx of buffalo (*Bubalis*

*bubalis*) at this site and this resulted in some grazing and numerous tracks through the understory. The large number of buffalo, due to their potential aggressive nature, made access difficult at times. In 2003 I also observed a large increase in weed density (*Xanthium* primarily) at this site.

Two sites (GNP1 and GNP2) were located 1.2 km apart within Gregory National Park (hereafter GNP), established in 1986 and covering approximately 13,000 km<sup>2</sup>. The first site (GNP1) was located at the Victoria River bridge (15.61972, 131.12902) and is adjacent to a roadhouse with a large campground. A narrow dirt track leads down to the river but is not used often and there was very little damage to the *Chionachne* from visitors. This is a favorite area for birdwatchers to look for the PCFW, but most choose to look for the birds from the highway and there was little disturbance to the birds or their habitat.

The second site within the national park (GNP2) was located at the Victoria River Access road (15.63189, 131.13225) where a 4-wheel drive boat-ramp was located, also on the western bank of the river. Recently (2005) this has been converted to a 2-wheel drive access area although great care was taken not to disturb the *Chionachne* habitat during construction and some stands that would have been destroyed were moved to a previously open area. Both sites were located in close vicinity to each other and had good access throughout most of the year. These two sites were chosen as they differed in *Chionachne* stand width and because their proximity to one another made it logistically feasible.

At both these sites, there was some buffalo and pig (*Sus scrofa*) activity with buffalo numbers increasing toward the end of the study. Nevertheless, grazing impact at these sites was not evident although some disturbances from these species (clearings and tracks) were present. Weeds (in particular *Xanthium* and *Ricinus*) were present at both sites in moderate abundance.

Wild passionfruit (*Passiflora foetida*) was also found in moderate abundance but only during the wetter time of year (November through April). Erosion was low to moderate at both the sites within the park but was particularly evident along the bank after the two big floods experienced in 2002 and 2003.

Table 2-1. Relationship between parent material, soils and topography in the VRD. (Kraatz, 2000).

Parent Material	Soils and Topography
Basalt	*red earths on sloping terrain *cracking clays (vertisols) on lower to flat slopes *rock outcrop and/or very stony surfaces common
Limestone, dolomitic and calcareous sediments	*earth and yellow earths on well drained slopes *cracking clays (vertisols) on poorly drained, lower slopes
Sandstone and calcareous sandstone	*sandy red earths *some yellow earths and lateritic podzolic soils
Non-calcareous shales (northern VRD)	*yellow earths *soils also prevalent on floodplain deposits
Calcareous sedimentary rock or basalt	*clays dominate alluvial plains

Table 2-2. Dominant vegetation in the VRD (data from Wilson *et al.*, 1990)

Dominant Vegetation	Common Tree Species	Common Grass Species
Eucalyptus woodland	<i>E. tectifica</i> <i>E. terminalis</i> <i>E. tretrodonta</i> <i>E. miniata</i> <i>E. polycarpa</i>	<i>Sorghum</i> <i>Chrysopogon fallax</i> <i>Sehima nervosum</i> <i>Plectracne pungens</i>
Low mixed species woodland	<i>E. pruinosa</i> <i>Terminalia arostrata</i> <i>Lysiphyllum cunninghamii</i>	<i>Eulalia aurea</i> <i>Chrysopogon fallax</i> <i>Aristida</i> sp. <i>Dicanthium</i> sp.
Low open woodland	<i>E. brevifolia</i> <i>E. dichromophloia</i> <i>E. miniata</i> <i>Lysiphyllum cunninghamii</i>	<i>Triodia pungens</i> <i>Plectrachne pungens</i>
Grassland		<i>Astrebla pectinata</i> <i>Chrysopogon fallax</i> <i>Dichanthium fecundum</i> <i>Astrebla</i> sp.

Table 2-3. Characteristics of *Chionachne cyathopoda* and *Mnesithea rottboellioides* (van Doorn and Low Choy, in prep.).

Species	Habitat	Soils	Gross Plant morphology	Leaf attachment	Reproductive (Inflorescence) morphology	Reproductive (Inflorescence) location
<i>Chionachne cyathopoda</i>	Banks and tributaries of large Rivers	Variety of soils from sand to loam to clay soils	Rhizomatous perennial with stems branching and intertwining to 4 m high	Throughout length of stem  Relatively small	Robust, enclosed by a prominent sheath  Lower half of functionally female flowers upper half of functionally male flowers	Terminal or in leaf axils
<i>Mnesithea rottboellioides</i>	Grassland / woodland associated with seasonal water (including creeks / wetlands perched on sandstone plateaus)	Generally sandy loam soils	Tussock perennial with leaves arising from base to 1 m high	From base of plant  Large forming bulk of plant	Fine, in groups near the end of a long stem  Sheath subtending flower groups not prominent	Stem exerted from the plant base commonly to 2 m

Table 2-4. Tenure and yearly conditions of field sites.

Field Site	Land Use	Land Tenure	2001 Site Condition	2002 Site Condition	2003 Site Condition
Coolibah Crocodile Farm (COOL)	Small enterprise	Freehold- private	Grazing-none Weeds-low Erosion- med	Grazing-high Weeds-med Erosion-med	Grazing-low Weeds-med Erosion-high
Dashwood Crossing (DASH)	Pastoral	Pastoral Lease	Grazing-med Weeds-high Erosion-high	Grazing-med Weeds-high Erosion-high	Grazing-low Weeds-high Erosion-high
Fitzroy Station (FITZ)	Pastoral	Aboriginal (pastoral lease 02-03)	Grazing-none Weeds-low Erosion-low	Grazing-low/med Weeds-med Erosion-low	Grazing-low Weeds-med Erosion-med
Gregory 1 (GNP1)	Conservation	Northern Territory Government	Grazing-none Weeds-med Erosion-low/med	Grazing-none Weeds-med/high Erosion-low/med	Grazing-low Weeds-med/high Erosion-low/med
Gregory 2 (GNP2)	Conservation	Northern Territory Government	Grazing-none Weeds-med Erosion-low/med	Grazing-none Weeds-med/high Erosion-low/med	Grazing-none Weeds-med/high Erosion-low/med

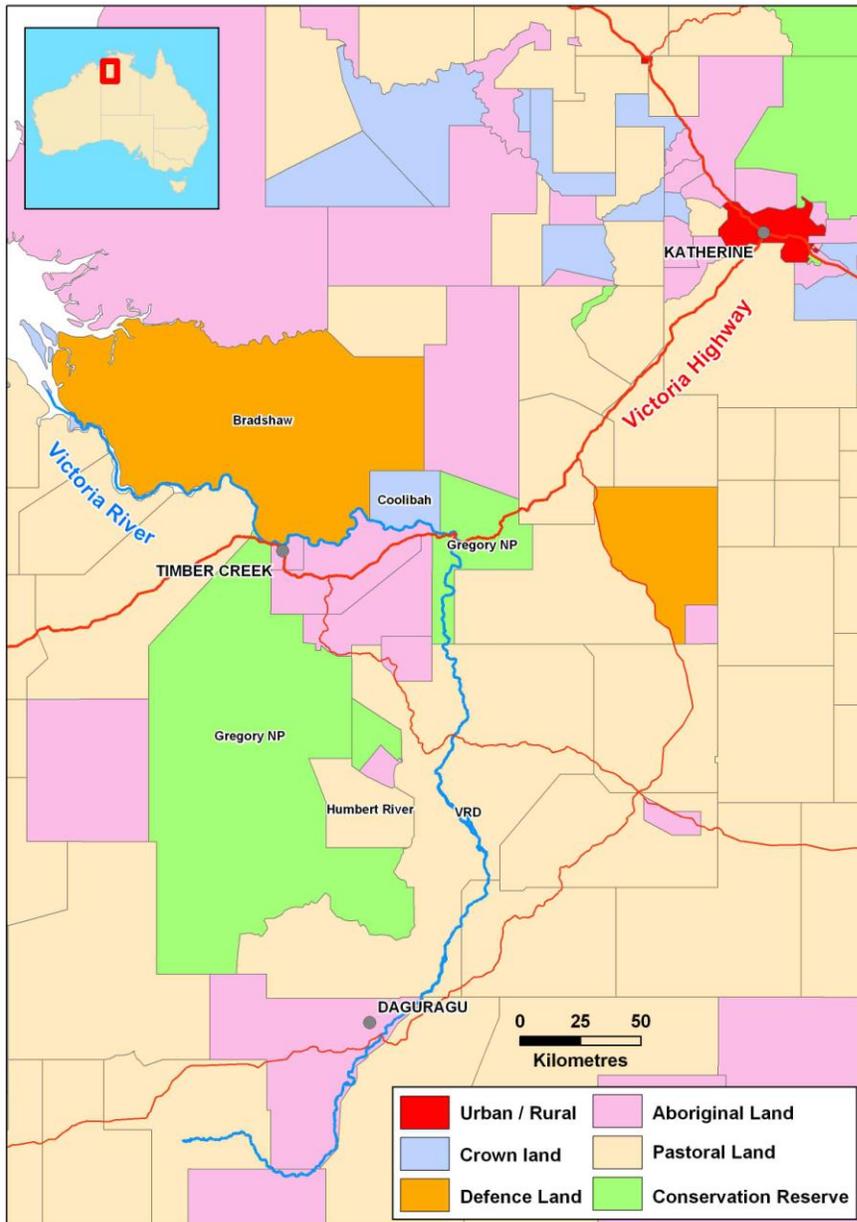


Figure 2-1. Location of the Victoria River District and associated land tenures. Note that Coolibah is under a perpetual pastoral lease.

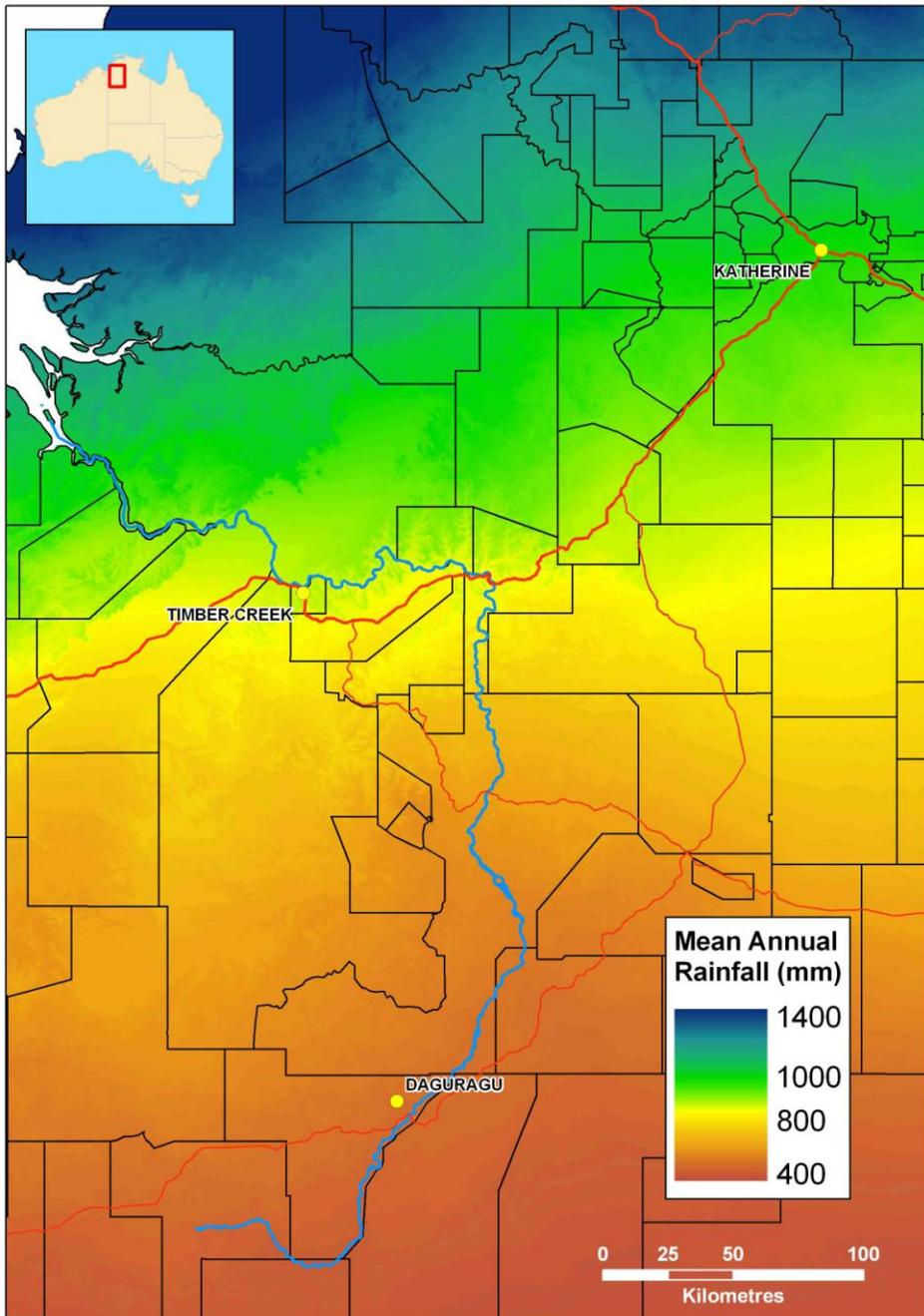


Figure 2-2. Mean annual rainfall in the study region.

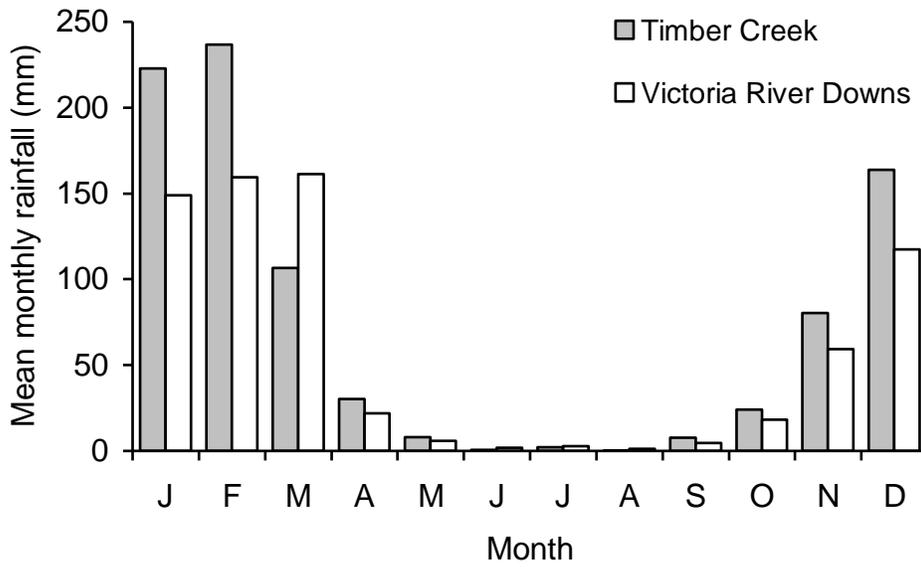


Figure 2-3. Mean monthly rainfall in Timber Creek (1981-2004) and Victoria River Downs (1885-2004). Data from the Australian Bureau of Meteorology, 2005.

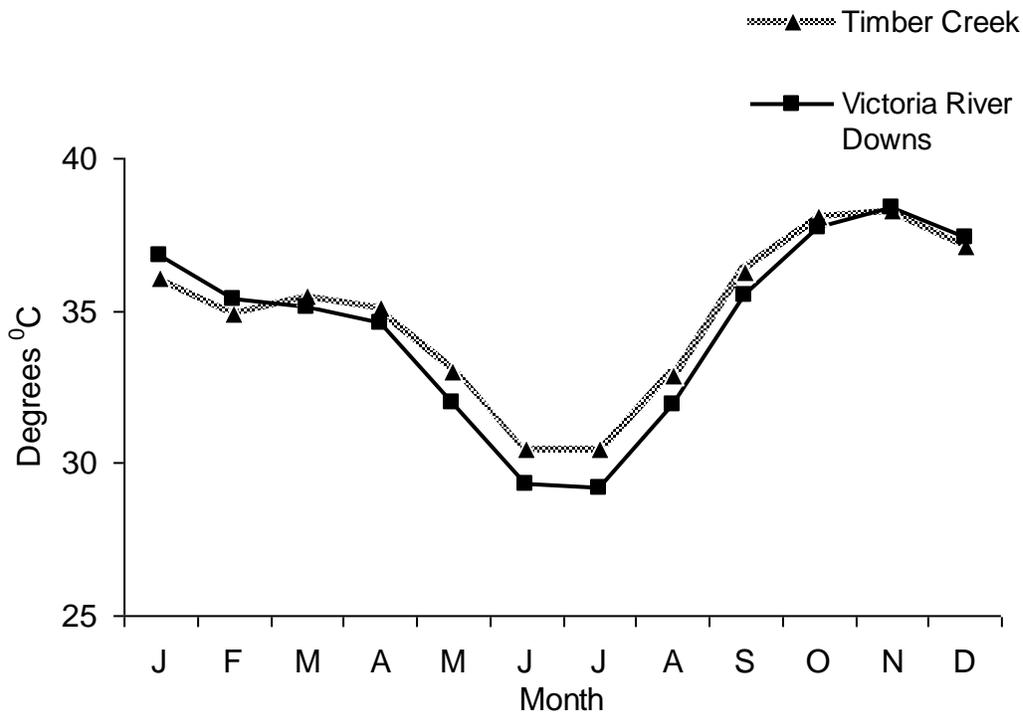


Figure 2-4. Mean daily maximum temperature Timber Creek (1981-2004) and Victoria River Downs (1885-2004). Data from the Australian Bureau of Meteorology, 2005.

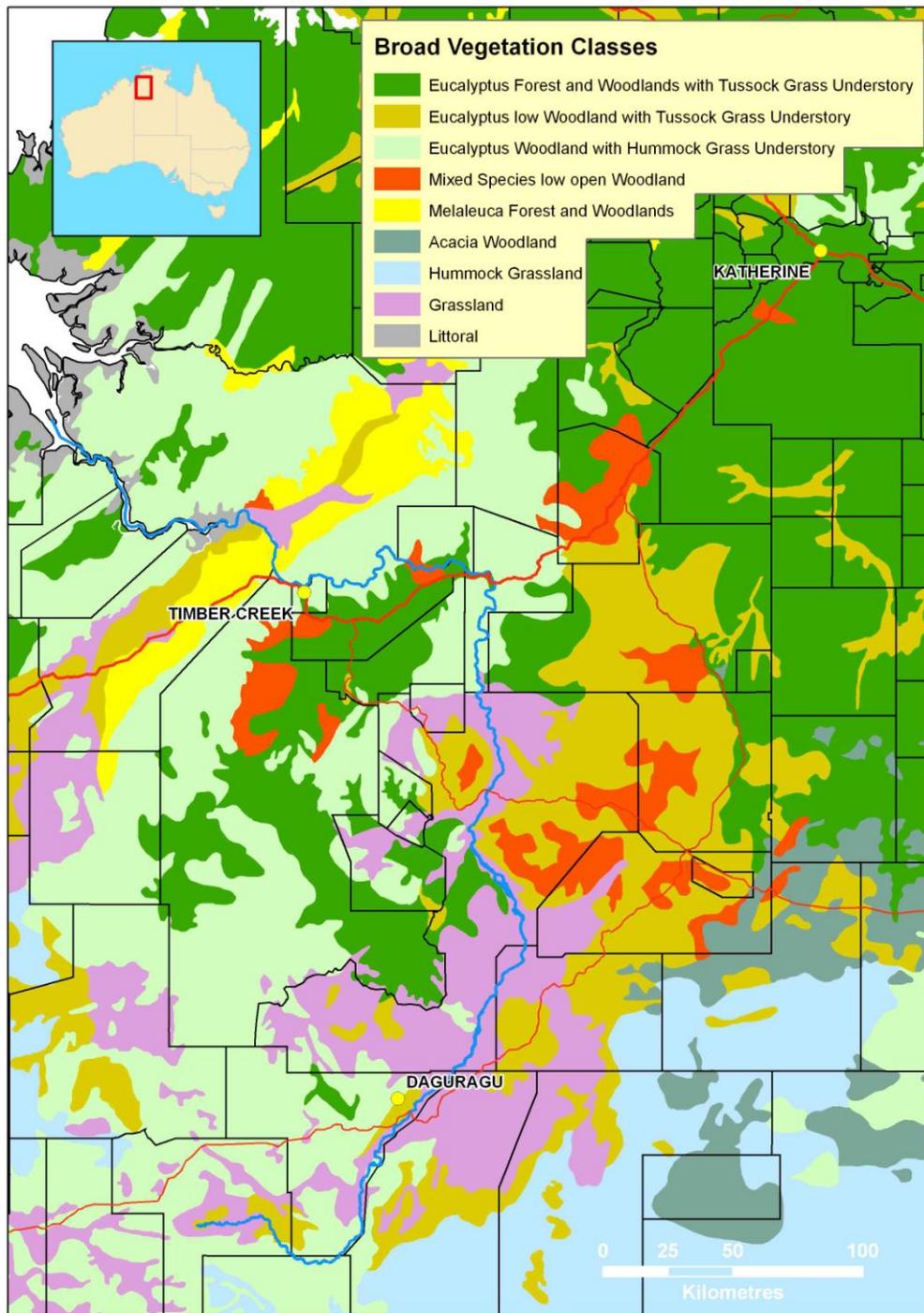


Figure 2-5. Broad vegetation classification of the study region.

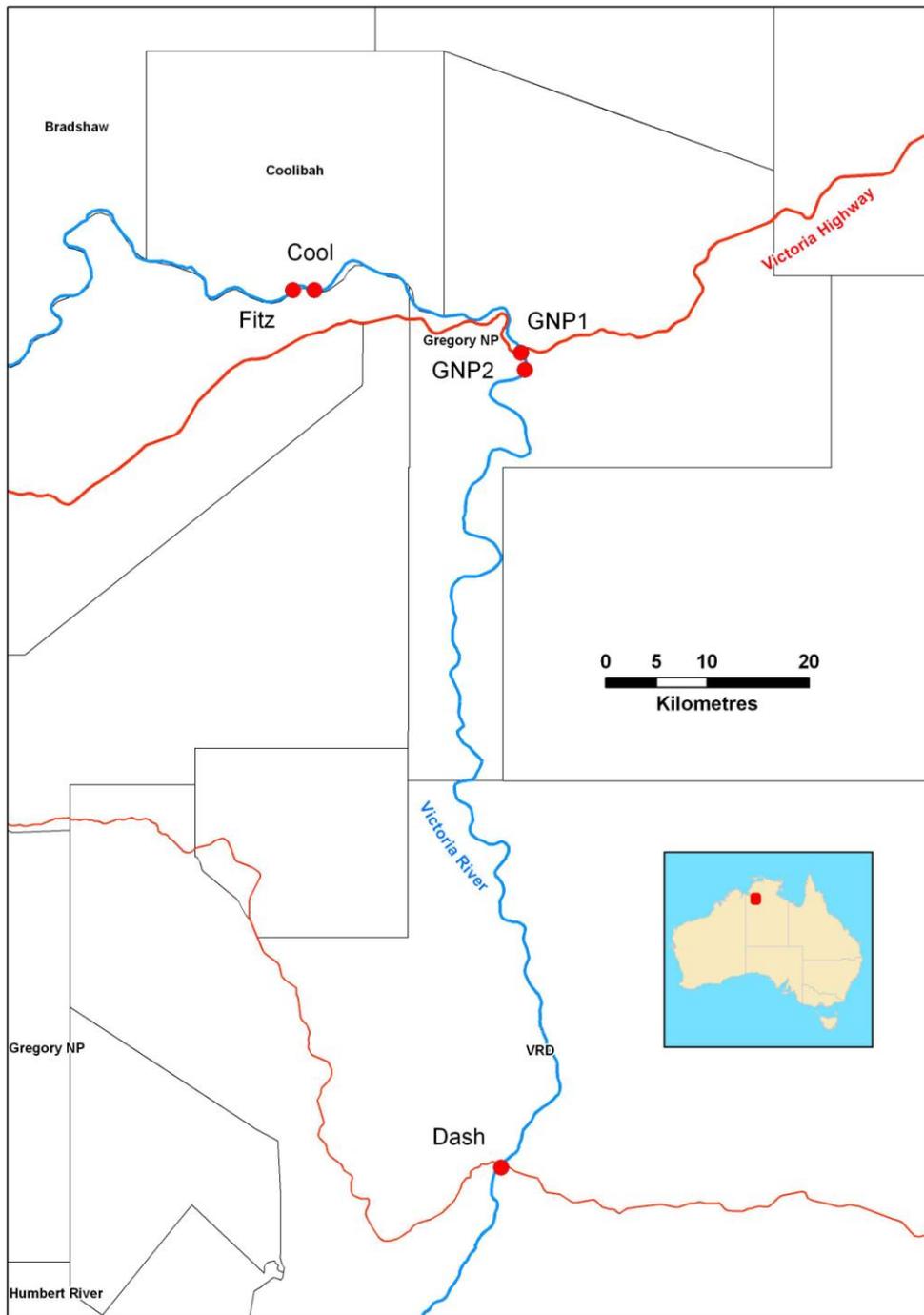


Figure 2-6. Location of study sites. Fitz = Fitzroy Station, Cool = Coolibah Crocodile Farm, GNP1 = Gregory National Park site 1, GNP2 = Gregory National Park site 2, Dash = Dashwood Crossing (Victoria River Downs Station). Note that indicated boundaries represent land tenure.

## CHAPTER 3 ECOLOGY OF THE PURPLE-CROWNED FAIRY-WREN

This chapter provides a description of the ecology of the PCFW, in the grass dominated riparian habitat in the VRD. This chapter also provides a comparison to the single in-depth previous study conducted in a *Pandanus*-dominated habitat. The ecology of the PCFW described in this current chapter provides an important foundation for the two subsequent chapters.

### **Introduction**

Many Australian passerines are characterized by their longevity, small clutch size, high incidence of multiple broods and low annual productivity (Fry, 1980; Woinarski, 1985; Yom Tov, 1987; Russell, 1989). The incidence of cooperative breeding is also high in Australian passerines (Rowley, 1969; Russell, 1989), a life history trait that has theoretically and comparatively been linked to high adult survival (Hardling and Kokko, 2003). The Maluridae, an old endemic family that has been distinct for 35-40 m.y. (Sibley and Ahlquist, 1985), exhibits many of these characteristics, and has been the subject of many landmark studies that have helped define, distinguish and interpret the biology of Australian birds in general (Rowley *et al.*, 1991; Mulder, 1997; Cockburn *et al.*, 2003).

Within the malurids, most such studies have been conducted on species occurring in the temperate south of the Australian continent. This bias gives some uncertainty to the extent to which biological traits for the genus as a whole can be generalized. In particular, there has been only one relatively brief previous study of the largest species in the genus, the PCFW, and that study (Rowley and Russell, 1993) was restricted to one site. Given that this species has a high degree of habitat specificity and occurs in a monsoonal climate contrasting starkly to that in temperate Australia, aspects of the biology of this species may exhibit some characteristics that differ from that currently regarded as typical for the genus.

Some previous long-term studies (Rowley *et al.*, 1991; Brooker and Brooker, 2001) of temperate-zone malurids (e.g. *M. splendens*, *M. pulcherrimus*) have shown how some life history characteristics (i.e. longevity and multiple broods) show some flexibility in response to variation in local conditions. It has been widely reported elsewhere that rainfall, for example, has been shown to influence productivity for many avian species as breeding is often influenced by food resources tied to the amount of precipitation (Lack, 1968; Perrins, 1970). Several *Malurus* species (*M. coronatus*, *M. cyaneus*, *M. pulcherrimus*) are known to reduce clutch size or abstain from breeding altogether following periods of low rainfall (Nias and Ford, 1992; Rowley and Russell, 1993; Brooker and Brooker, 2001).

Many *Malurus* species have annual adult survival rates around 70%, hence, the strategy of surviving the bad years, such as low rainfall years, without breeding is a viable option (Rowley and Russell, 1997). Conversely, during favorable years, females will often produce multiple broods and breeding season may be extended, enabling them to exploit the favorable conditions and potentially compensating for bad years (Nias and Ford, 1992; Brooker and Brooker, 2001; Rowley and Russell, 2002).

In the wet-dry tropics of northern Australia, inter-annual variation in rainfall can be very high (Ridpath, 1985; Taylor and Dunlop, 1985) resulting in a change of the breeding behavior of PCFWs (Rowley and Russell, 1993). Indeed, Rowley and Russell (1993) found no evidence of breeding of PCFW following two wet seasons marked by low rainfall. In addition to climatic variation, changes in habitat attributed to disturbances can also trigger changes in breeding behavior. For example, *M. elegans* displayed delayed breeding after a fire and productivity was lower the following year (Russell and Rowley, 1998)

This chapter describes aspects of the life history and behavior of PCFWs at a set of sites in the VRD. It specifically considers the following key issues.

- What are the basic characteristics of breeding biology for this species in the VRD?
- To what extent do these characters vary between years?
- What is the diet and foraging behavior of this species in my study area?
- To what extent do the breeding biology and other characteristics reported here for this species in the canegrass of the VRD contrast with those reported in the only previous study of this species, in a *Pandanus*-dominated habitat in the east Kimberley? And, to the extent that there may be any marked differences, what environmental factors would contribute to such distinctions?

Given that my study substantially amplifies the knowledge base of the PCFW, the discussion in this chapter also considers the extent to which the breeding biology of this species falls within the characteristics typical for the genus as a whole.

There are some important caveats that constrain the extent to which these questions could be addressed in my study, and the interpretation of the results obtained. My study occupied three field years, so there were limited opportunities for considering inter-annual variation in environmental conditions and bird responses. Further, in contrast to the previous study of Rowley and Russell (1993), rainfall conditions over the three years of my study were relatively similar. My skill at nest location and other factors increased over the course of the study, so results from the first year are more fragmentary than for subsequent years. Comparison among years was also hampered by the loss of one key study site for one year, because of unpredicted social issues with one landholder. I also acknowledge that my information on diet is limited: in the extremely dense understory of this area, there are few opportunities for prolonged observation of this furtive species.

Comparison with the previous study of PCFWs by Rowley and Russell (1993) is an important component of this chapter. The ecological characteristics determined from only one

study site form a fragile base for characterizing biological features and conservation management conclusions. Although both my and Rowley and Russell's (1993) sites are narrowly riparian, there are significant differences in management interpretation. For example, *Chionachne* is more readily disturbed by cattle and is a less fixed structure than *Pandanus*; *Chionachne* may extend further from riverbank and hence dispersion of PCFWs is less linear. There are some interpretational issues associated with the direct comparisons of these 2 studies: Rowley and Russell's (1993) publications don't necessarily contain full data so statistical comparisons are constrained. In addition, due to structural differences between the riparian habitats some comparisons are also hindered. For example, they reported PCFW density as a linear measure in comparison to the area measurement I used.

This chapter does not consider variation in biological characteristics among my study sites, associated with varying levels of disturbance. That topic is the subject of Chapter 4.

## **Field Methods**

### **Terminology**

I use the term "pair" to define the breeding female and dominant male in a group. Any additional adult birds were termed "helpers". Juvenile birds were those identified as being progeny of the year and hence less than one year in age (1-). "Eclipse" plumage refers to the characteristic non-breeding plumage of adult males. Adult birds were more than one year in age (1+). Based on plumage, adult birds could be categorized into the following age groups; female (2-), female (2+), male (2-), male (2+), male (3+) (Rowley, 1988).

To identify the breeding pair if a group was made up of more than a single pair, the female with brood patch was identified as the breeding female and the eldest male as her partner. The identity of a breeding female was also corroborated when she was frequently fed by males in her group. On a few occasions males within a group were almost identical in appearance and in

these instances I considered the senior male to be the one that was most often found performing a duet (singing together) with the female. However, it is important to note that “pair” is a difficult concept to unambiguously apply to cooperative breeders, and that even in cases where a dominant and apparently paired male and female share a territory, other studies have shown that paternity is not necessarily straightforward (the females often sneaking out to mate outside the territory with a more distant male, Double and Cockburn, 2000).

I use the term “group-year” to denote various aspects of a family group over a one-year period (i.e. productivity, composition, size) (Rowley and Russell, 1997). Only adult birds (1+) were counted as being part of a group. I use the term “divorce” in instances where a breeding pair did not remain together in the absence of the death of either bird (Rowley and Russell, 1993).

Productivity was measured as the number of independent juveniles produced per group (i.e. breeding female) during the breeding season (March-September). Nests were termed as “unsuccessful” if they failed to produce any fledglings and were classified as been unsuccessful in either the egg or nestling stage. Nests termed as “successful” were those that produced at least one fledgling to independence (Rowley and Russell, 1997).

I measured annual survival of adults as the proportion of birds present at the beginning of each breeding season that were still present at the beginning of the following breeding season.

### **Captures and Banding**

I captured individuals with mist nets from 2001 through to 2003 at the five previously identified field sites (Coolibah, Dashwood, Fitzroy, GNP1 and GNP2- the two sites within Gregory National Park).

I placed mist nets within known territories, and used a playback system to attract birds to the nets. At times only one net was used, but in open areas or where birds had proven hard to

catch, multiple nets were used to increase the probability of capturing all group members. Nets were placed away from territory boundaries to avoid capturing multiple groups simultaneously or disturbing neighbouring groups. When possible, I placed nets along drainage lines and other open areas to avoid damaging wren habitat.

Once captured, I removed birds and placed them individual cotton holding bags before processing. Nets were never left unattended and birds were customarily removed from the net within one to five minutes of capture. Although the aim was to capture the entire group, individual birds could be extremely wary and fledglings would often avoid approaching the nets altogether.

I focused efforts on capturing birds that were un-banded. In instances where all members of a group were already banded, groups were not purposely recaptured as the birds were easily identified by their color bands. I could not band birds at Fitzroy Station in 2003 due to restricted access and social issues.

Birds were banded according to the Australian Bird Banders Association guidelines (1989). Each bird was banded with numbered aluminum band (13 mm, size 02) and unique combination of three colored plastic bands for subsequent visual identification.

### **Morphology and Plumage**

Based on plumage, each captured bird was sexed and age (category) and molt condition for each bird was recorded. During subsequent visits to field sites any observed banded birds were recorded and any changes in plumage noted. For each captured bird I recorded the following data: weight, wing length (flattened folded right wing), tail length and tarsus length.

There were some occasions where morphological measurements were not obtained, a photograph was not taken, or film was damaged due to excessive heat resulting in missing data. Missing morphological data could be attributed to one the following; birds (primarily breeding

females) became stressed or were suspected of carrying an egg and were only banded, birds escaped before all data were collected, or a measurement was overlooked at the time of capture. Photographic quality improved once a digital camera was used from 2002 onward.

Differences between sexes (and between adults/juveniles) in dimensions and weights were assessed with t-tests.

As females may lay up to four clutches in one season, physical condition may deteriorate through the season. I compared female weights between the beginning (May and June) and the latter part (July and August) of the breeding season to ascertain any average difference in body weight. For seasonal comparisons of weights, (three capture seasons), I kept samples independent by using only one measurement per female in the event of a recapture. I compared differences in weights using t-tests.

### **Group Size and Composition**

I determined group size and composition by two methods. The first method was to identify group size and composition whilst mist-netting. Although the entire group was not always captured, the close proximity of other birds belonging to the same group made it fairly easy to identify them during mist-netting attempts. Secondly, throughout the field season, groups would be identified and followed regularly (during nest searches and foraging observations). Any birds encountered were recorded. Since most groups were followed on at least two occasions, group composition was well known.

I compared group size across the three years of my study using a Kruskal-Wallis test. I used a Chi-Square ( $\chi^2$ ) test to determine if sex ratios varied from parity. I also compared the frequency distribution of group sizes, (total no. birds per group), in my study area with those recorded by Rowley and Russell (1993) for Drysdale River area, using a Kolmogorov-Smirnov test

## **Territory Shape and Size**

One of the primary techniques for determining territory boundaries was by recording any territorial singing, either individually or as a duet. Birds were particularly active in territorial singing at the beginning of the breeding season and much less so at other times of the year when territories became less defined.

At the beginning of the breeding season adjoining family groups would often respond to each other with song to mark boundaries. Further, groups rarely wandered far into adjoining territories during the breeding season. By repeatedly recording positions of singing birds, territory boundaries were often clear. Additionally, birds were followed regularly to monitor breeding and record foraging observations, making it was possible for me to determine territory boundaries of groups. During mist netting, territorial boundaries became clear when two adjacent groups were attracted to the net but rarely crossed into each others territories.

I constructed territory maps during the breeding season 2002 when the most time and resources were available and territories were well known. During the 2002 season, family groups were followed extensively and any territorial behavior recorded and marked using a GPS (Garmin GPS II Plus <15m). Data points were plotted as minimum convex polygons on Arcview 3.2 to determine territory shape and size and were subsequently mapped using ESRI ARC GIS 9.1. Territory sizes among sites were compared using a Kruskal-Wallis test.

It was not possible to compare territory size for birds in my study area with those reported by Rowley for birds in the Drysdale River area, because Rowley reported only (streamside) length for the territories that he observed and not the width.

## Foraging Behavior

I recorded foraging behavior opportunistically by following individual birds as long as possible. The time when an individual was first seen was recorded as well as each time a foraging observation was made until the last foraging behavior was observed.

The following data were collected for each observation:

sex and identity of individual;

foraging behavior following Rowley and Russell (1997), as either:

- hawk = flying bird takes flying prey,
  - hover = bird remains stationary in air before or during taking prey,
  - glean = bird is perched whilst taking prey,
  - probe = bird inserts beak into substrate to obtain prey,
  - pounce = bird flies down from perch to take prey on ground, or
  - snatch = bird takes prey from substrate during flight,
- foraging substrate, as either
- *Barringtonia* (freshwater mangrove),
  - *Chionachne* (canegrass),
  - *Xanthium* (noogoora burr),
  - Bareground,
  - Bareground w/ debris,
  - tree species, or
  - other (as identified); and
- prey item.

Although I followed birds as long as possible, the density of the *Chionachne* habitat made it very difficult to follow any individual for an extended period of time. It was even harder to record prey items since their small size limited visibility and they were quickly ingested.

I used Chi-Square ( $\chi^2$ ) tests to compare the foraging observations for males and females to determine differences in foraging substrate. I also used Chi-Square ( $\chi^2$ ) tests to determine substrate foraging preferences based on vegetation abundances. It was not possible to make rigorous quantitative comparisons of the foraging behaviors recorded in my study with those reported by Rowley and Russell (1993) in the Drysdale River area, because they used very different methodologies (analysis of fecal pellets), however broad qualitative comparisons are reported.

### **Breeding Biology**

In each of the three years of my study, rainfall during the wet season was above average and the primary habitat of the PCFW was flooded around February each year.

From the beginning of each field season (March/April/May) I followed family groups at each site to identify any nesting behavior. The recession of flood waters and access to field sites determined the start of observations at each sites. Some sites were accessible later than others depending on road conditions (i.e. Coolibah and Dashwood Crossing).

The most common behaviors that indicated breeding were a female carrying nesting material or individual birds carrying prey items to nest sites. Females begging for food or being fed by other members of the group were other indications that they were breeding or may soon breed. On several instances breeding pairs were observed soon after copulation and once during copulation. These birds were then followed to determine the specific nest location. After identification of a nest site, I determined the status of the nest when birds were not in attendance in order to minimize disturbance.

Each nest was monitored every four to six days (when possible) and nest status was determined (stage of construction and nest content). Due to the distance between sites, this was sometimes impossible and nests were checked less frequently. Because checking nests at greater frequencies could increase the already high predation rates, by leaving scent trails or being observed by potential nest predators, I arbitrarily decided to check nests every six days. During nest checks, I took different routes to the nests and spent as little time as possible at nests. I also routinely checked for the presence of any potential nest predators (e.g. Torresian crow *Corvus orru*, pheasant coucals *Centropus phasianinus*, *Varanus* sp. etc.) before approaching a nest. Nest checks were conducted when group members were not in attendance when possible. If a female was incubating at the time of inspection, I would leave the site to avoid flushing the female and potentially increase the risk of nest abandonment. When nestlings were aged around six days, they were banded with a numbered aluminum band and three color bands as per adult birds.

Once a nest was no longer in use (after nest failure or successful fledging), I collected nest measurements. Nest measurements included; GPS location, substrate, entrance orientation, % cover (vegetation) within 1 m diameter of the nest, cover description (plant species), nest base height above ground and nest entrance height above ground. There were some instances where nest measurements were not possible because nests had been completely destroyed and/or were inaccessible due to the presence of feral animals (buffalo or scrub bulls) that may have endangered observers. On only two occasions volunteers, could not locate previously-found nests.

Clutch size and productivity were compared between years, using a Kruskal-Wallis test.

## **Annual Survival**

I only included banded birds for annual survival rates because the identity, and therefore survival, of un-banded birds was unknown. Note that this assessment may be confounded by movement of individual banded birds beyond the territory they held the previous year. Where such birds were re-sighted elsewhere, these were included in the survival estimates.

Survival rates for juveniles could not be adequately estimated as few nestlings were banded due to high predation rates, in addition to low numbers of juveniles captured in nets. Additionally, the primary habitat of wrens was continuous at all sites, thus increasing the likelihood many juveniles may have dispersed into adjacent areas outside of my study sites before they could be located and banded.

Median survival rates between years, 2001 to 2002 and 2002 to 2003, were compared with a Mann-Whitney U test.

## **Results**

### **Captures and Banding**

A total of 177 PCFWs was captured and banded during the three field seasons comprising 54 females, 66 males and 25 juveniles less than one year of age (Table 3-1). In addition, nine nestlings (three clutches) were banded and 32 adult wrens were recaptured. During the first (2001) field season 22 birds were banded outside of the five field sites (see Chapter 2), mainly due to restricted access to some sites the following field season (2002). Wrens were banded only at the five identified field sites during 2002 and 2003. In addition to the birds banded from 2001 through 2003, a small number of birds (six) were captured at two sites in 2000.

There was no recorded mortality of birds due to captures. Only three (female) birds became stressed, the weights of two of these females suggested that they were carrying an egg; these females were released as soon as possible and located on subsequent surveys. Two minor

injuries were recorded during my study. On one occasion an adult male suffered a small scratch on the tarsus when a *Xanthium* seed became lodged in the mist-net. On the second occasion an adult male received minor injuries attributed to a collared sparrowhawk (*Accipiter cirrhocephalus*) attempting to take the bird out of the net. Both birds survived were located on subsequent surveys.

### **Morphology and Plumage**

I found morphological differences between males, females and juveniles (summarized in Tables 3-2 and 3-3). The average male was significantly larger than the average female in mass (males  $11.1 \pm 0.6$ g, females  $10.6 \pm 0.7$ g,  $t = 3.47$ ,  $P < 0.01$ ), wing length (males  $54.0 \pm 1.7$ mm, females  $51.7 \pm 1.5$ mm,  $t = 7.76$ ,  $P < 0.001$ ), tail length (males  $71.7 \pm 5.4$ mm, females  $68.7 \pm 6.5$ mm,  $t = 2.79$ ,  $P < 0.01$ ) and tarsus length (males  $21 \pm 1.1$ mm, females  $20.2 \pm 1.3$ mm,  $t = 3.26$ ,  $P < 0.01$ ).

Juveniles were significantly smaller than adult males for all measurements with the exception of tail length. Juveniles were not significantly smaller than females except in weight. Interestingly, juveniles had longer tails on average ( $75.2 \pm 4.7$ mm) than either males ( $71.7 \pm 5.4$ mm:  $t = 2.98$ ,  $P < 0.01$ ) or females ( $68.7 \pm 6.5$ mm:  $t = 5.06$ ,  $P < 0.001$ ), despite a high degree of variation. The tails of juveniles were very short when they fledged and became progressively longer until the first molt. Many birds were molting during capture and the high degree of tail length variation is most likely attributable to molt cycle.

Females were heavier at the beginning of the breeding season (weight in May-June was  $10.9 \pm 0.8$  g ( $n = 20$ ) than later in the season (weight in July-August was  $10.5 \pm 0.7$  g,  $n = 40$ :  $t = -2.07$ ,  $P < 0.05$ )

Breeding males were easily identified by facial plumage (Fig. 3-1). Males have a characteristic purple crown with a black oblong spot in the centre, wide black facial mask that

extends around the nape and black bill. Adult females have grey crowns (ranging from a dark bluish-grey to grey with varying amounts of brown), chestnut ear coverts, complete white eye ring, distinct white eye brow, and brown bill (Fig. 3-2). Tails in both males and females are blue sometimes tinted greenish. The upper bodies of males and females are cinnamon brown and the lower (chin, belly and breast) are cream with varying shades of buff.

Eclipse males could be identified by their grey (sometimes partly brown) crowns, and grey to blackish ear coverts (Fig. 3-3). Some males kept their breeding plumage throughout the year and a few males were still in eclipse plumage well into the breeding season although were known to be breeding based on the fact that they were the senior male or only male in the group.

Juvenile birds (less than one year of age) were identifiable by their lack of male or female plumage (Fig. 3-4). Juveniles have brown crowns with none or very little grey, chestnut to brown ear coverts, brown (often light brown) bills, incomplete white eye circles, and lacked the obvious white eye-stripe found in adult females (although some white may be present). In addition, juveniles often had longer tails (Table 3-3) and could sometimes be identified by their behavior. As juveniles matured, younger males were easier to identify than females as the former would acquire a few grey to blackish feathers in the ear coverts, purple or black feathers on the crown and a blackish bill. However, care had to be taken to not confuse juvenile males starting to acquire adult plumage with eclipse males or males molting. In general, I knew which groups had juveniles present and the dominant male within that group so identifying young males was easier.

Confusion between juveniles and females can also occur and plumage varies greatly among individuals. Although adult female PCFWs usually have a grey crown, I found this to be highly variable, with numerous females displaying varying degrees of grey and brown plumage

on their crown. The degree of grey plumage appears to be linked to moult and not age (e.g., one female was at least six years old but yet her crown had a substantial amount of brown). Another adult (2+) female was first captured in 2001 and had a predominantly grey crown, but when she was recaptured in 2002 a lot more brown was present and finally in 2003 her crown was a distinct jet grey. In addition to head coloration, breeding females could often be identified by bent (curved) rectrices from incubating, thus further distinguishing females from juveniles.

### **Group Size and Composition**

PCFW live in small family groups ranging in size from two to five birds, consisting of a breeding female and senior male and commonly included additional helpers (Table 3-4). The most frequently recorded group composition (in 50% of cases) was a single breeding male and female. The total number of males in these groups (88 group-years) was 137, and of females 93: i.e. a sex ratio of 1:1.47, this differs significantly from parity ( $\chi^2 = 4.611$ , DF = 1, P < 0.05). However, groups with one additional helper were also common (41%). Mean group size was  $2.6 \pm 0.7$  (median 2.5) based on 88 group-years. Towards the end of the breeding season the number of individuals in a group could rise up to eight or more including progeny of the year (note that these latter were not included in this tabulation).

Helpers in groups were predominantly males, and ranged in number from one to three; 44.3% of group-years had one or more male helpers. Female helpers were rare and only 5 (5.7%) group-years had a female helper, and none of these were present for the duration of the breeding season. Male helpers (usually progeny of the previous year) frequently stayed in their natal territory for consecutive years. During my study, 16 males remained helpers for at least two consecutive years.

Pair bonds were often maintained over consecutive years with replacement of individuals of a pair occurring after either death or divorce. I recorded 10 breeding pairs that remained

together for three consecutive breeding seasons and three breeding pairs remained together for two breeding seasons. Divorce was recorded three times and all six birds involved stayed within three territories of where they were first recorded. It is likely that more breeding pairs remained together for consecutive breeding seasons but as some birds were not banded it was impossible to determine the fidelity of those birds.

### **Territory Shape and Size**

I mapped 30 territories in 2002, nine at Coolibah, five at Fitzroy, five at GNP1 and 12 at GNP2. Territories at both Coolibah and Fitzroy were mapped before cattle arrived. Territories were not mapped at Dashwood due to time constraints. Territories are best described as polygons with an average size of 0.41 ha (4,135m<sup>2</sup>). The shape of territories was highly dependent on width and distribution of the *Chionachne* and some sites had multiple territories perpendicular to the river (Figs. 3-5, 3-6). Territories were smallest at Coolibah (0.26 ha) and Fitzroy (0.43 ha) and larger within the two national park sites, GNP1 (0.55 ha) and GNP2 (0.45 ha), the difference in mean territory size was statistically significant ( $H = 8.62, P < 0.05$ ).

### **Foraging Behavior**

I recorded 963 minutes (16.05 hrs) of foraging observations at various locations during the 2002 and 2003 field seasons, 350 minutes were recorded for females, 550 for males, and 63 for juveniles. The greater number of observations made for males can be attributed to a larger number of males being present at the study sites and greater ease of following males with their conspicuous plumage. The lower number of observations for juveniles can be attributed to the fact that tracking juveniles was more difficult as they were generally more timid and many had not yet been banded.

Wrens tended to forage in loose groups and I often observed juveniles following adults as the group moved through vegetation during the breeding season. On occasions where the

dominant male and female were busy re-nesting after having produced a successful clutch, juveniles were often found with helpers and on numerous occasions I observed helpers feeding juveniles. When birds were foraging as a group, contact calls could be heard as the birds moved through dense understory. Wrens tended to forage individually when feeding nestlings or juveniles.

Birds spent the most time foraging in canegrass *Chionachne* (60%), usually within 2 meters of the ground (Fig. 3-7). Interestingly, the introduced herbaceous weed, *Xanthium* (13%) was the second most common foraging substrate followed by freshwater mangrove *Barringtonia* (12%), a small tree 5-8 m in height, and bareground with debris (8%). Less time was also spent foraging in tree species other than *Barringtonia* (3%), bareground (2%) and other substrates such as the introduced weed castor oil *Ricinus* and the small native shrub *Flueggia*(2%). In total, these comprised only a small portion of time compared to other substrates. Comparisons between vegetation cover and foraging substrate indicated that *Chionachne* and *Barringtonia* were the preferred substrates ( $\chi^2 = 123.981$ , DF = 4, P < 0.001).

Overall, birds did occasionally forage in tall trees, but by far the majority of time was spent within 4 meters of the ground. Observations during floods also suggest that taller species (e.g. *Barringtonia*, *Eucalyptus* spp., *Ficus*) are important foraging substrates during floods when *Chionachne* is inundated.

There was a difference in foraging substrate between males and females ( $\chi^2 = 15.282$ , DF = 5, P < 0.01). Males showed a greater affinity for *Barringtonia* in comparison to females, whereas females showed a stronger affinity for *Chionachne* in comparison to males (Fig. 3-8). Although the sample size for juveniles was too small to analyze statistically, there were some notable differences in foraging preferences based on the limited sample size (n = 63 min).

Juveniles spent approximately the same percent of time foraging in *Chionachne* as adults (57%) but also favored bareground with debris (19%) and a lower preference for *Barringtonia* than either males or females (6%).

Of the 110 foraging behaviors recorded (Fig. 3-9), gleaning was by far the most common (60%) followed by snatching (18%). The following behaviors were recorded less frequently: pounce (8%), probe (7%), hover (4%) and hawk (3%).

On 27 occasions the prey item captured by a bird was identified (Fig. 3-10). Lepidopterans (butterflies and moths) were the most common prey item captured (48%), followed by Orthopterans (grasshoppers, crickets and katydids – 19%), Hymenopterans (ants, bees, and wasps – 19%), Hemipterans (bugs, aphids and hoppers – 7%), and Dipterans (flies and mosquitoes – 7%).

### **Breeding Biology**

I recorded a long breeding season of 22 weeks, the earliest month an active nest was recorded was in March (2002) and the latest a nest was found was in September (2003) with the majority of nests found in May (Fig. 3-11). No evidence of breeding was found from October to February.

During the three field seasons, 65 active nests were found; eight in 2001, 22 in 2002 and 35 in 2003. Of the 65 nests found, 24 were found at the GNP1, 24 at GNP2, 10 at Coolibah and 6 at Dashwood. Nest searches were not conducted at Fitzroy (see methods).

Purple-crowned Fairy-wren nests are large domed structures typical in the *Malurus* genus. In my study, only females built nests, but females were often accompanied by the dominant male or the entire group during the building process. A female often showed an affinity for a specific area within the groups' territory. This was true for successive clutches in a single year, and also for some females nesting in the same territory over successive years. Nests were also frequently

placed close to clearings (e.g. drainage areas, roads, and tracks), perhaps to provide a better escape route for females from predators. Nests situated on the edges of drainage lines (i.e. generally small erosion valleys caused by flooding) were also generally higher, as nests would be placed in *Chionachne* on top of the bank, and would be more difficult to access for some predators (i.e. non-avian predators).

Purple-crowned Fairy-wrens showed a strong affinity for *Chionachne* as a nesting substrate; 85% (55) of nests were located in *Chionachne*. The remaining nests were placed in *Mnesithea* 14% (9), another clumping riparian grass similar to *Chionachne*, and one single nest (1%) was found in a *Barringtonia* sapling. Nests were constructed of varying amounts of *Chionachne* blades (the principal nesting material), *Mnesithea* blades, strips of bark, and leaves with other small rootlets, twigs, and spider webs added (usually in the nest lining).

Nests were generally located low to the ground near the base of either *Chionachne* or *Mnesithea* grass clump. The average height of the base of the nest from the ground was 30.7 cm and the average height of the nest entrance was 40.1 cm above the ground (Fig. 3-12). Nest entrance orientation varied but predominantly faced east (73%). The amount of cover (vegetation) surrounding nests ranged from 30% to 90% with an average coverage of 64.7%.

Re-nesting was common after either failed or successful nests and females would lay as many as four clutches in any given breeding season. From 2001 to 2003, I found 13 (15%) groups that had at least two nests, 17 (19%) groups that had at least three nest and five (6%) groups that had at least four nests in a single season. Five groups (6%) of 88 group-years successfully fledged young from two clutches in a single year. All of these belonged to older females that were at least three years old.

Clutch size was determined for 49 of the 65 active nests. Eggs were faded white in color and had brown/reddish splotches predominantly on the rounded end. The mean clutch size was 2.9 ( $\pm$  0.4), with 88% of clutches consisting of three eggs, 10% of two eggs and 2% of one egg. Only females incubated and eggs were laid on successive days. When a female was incubating, the senior male would sometimes follow the female to the nest and wait in nearby vegetation. However, most of the time the female would approach the nests alone and rejoin the group at the end of each incubating session. On occasions when females were flushed, they consistently moved off the nest at the last moment.

I estimate incubation to be approximately 14 days. The nestling period lasted approximately 10 days in my study. Fledglings remained cryptic during the first week and stayed well hidden in dense undergrowth where they could often be heard begging for food as other group members approached. When potential predators were nearby, fledglings would be led away by group members. Further, all group members, including helpers exhibited decoy behavior in the presence of potential predators, performing “rodent-runs” (Rowley and Russell, 1997). The rodent-run was also exhibited when I approached a nest or when juveniles were captured in mist nets.

During my study, high nest predation rates were found. Of the 51 nests where at least one egg was present; 10% of the clutches did not hatch or were abandoned and 39% of clutches were failed at the egg stage (predation) before hatching, only 51% of the clutches hatched. Of the 26 hatched clutches, 19% failed at the nestling stage (predation) but 46% fledged young. The outcome of nine nests was unknown (i.e. it was not possible to confirm whether nestlings fledged or were victims of predation, however, no fledglings were found). The combined predation rate

for the 51 clutches was 51% with the possibility that the rate could be as high as 67%. I found no evidence of nest parasitism during the course of my study.

I was not able to identify predators for any of the nests with the exception of two where cats (*Felis catus*) were identified by paw prints and both nests were completely destroyed. Predation by avian species is often indicated by a hole at the back of the nest; one nest was found in this condition suggesting an avian predator. Complete destruction of nests was very rare and nests were usually found intact or with only minor disturbance to the nest entrance. Nests were usually found completely empty with no sign of egg shell meaning the eggs were taken whole. Potential predators, all of which were frequently observed at study sites, include blue-winged kookaburra *Dacelo leachii*, pheasant coucals *Centropus phasianinus*, Torresian crows *Corvus orru*, goannas *Varanus* spp., several snake species, and rats (including *Rattus rattus*).

In addition to the outcome data for the 51 nests with eggs described above, it was possible to determine which groups had fledged at least some offspring during the breeding season and calculate group annual productivity. Overall, the mean number of fledglings produced per group-year (i.e. per breeding female of 88 groups) was 1.07 based on the number of juveniles I encountered and witnessed fledging. Productivity was very similar among years and the mean number of fledglings produced per female was 1.05 in 2001 (n = 22), 1.09 in 2002 (n = 34) and 1.06 2003 (n = 32) (H = 0.21, P = 0.9).

It is quite possible that my productivity level is an underestimate as some juveniles may have dispersed or been predated before I could find them. For example, one group at Coolibah in 2003 fledged three young, however, each second day this group was visited the number of fledglings was reduced by one and no fledglings were left after six visits. Presumably, as these

fledglings were still dependent on adults and too young to disperse, they were taken by predators.

### **Annual Survival**

Adult annual survivorship was relatively high with 61% (68 of 112) adult banded birds surviving from 2001 to 2003. Survival rates were higher from 2001 to 2002 (70%) than from 2002 to 2003 (56%) ( $U = 358.5$ ,  $P < 0.05$ ). The lower survivorship rate from 2002 to 2003 can be attributed to the high number of birds lost from Coolibah, the intensely grazed site, during this time (71%) as discussed in detail in Chapter 4.

## **Discussion**

### **Ecology of the Purple-crowned Fairy-wren in The Victoria River District**

In my study of the PCFW in the VRD, average adult survival was 61%, mean clutch size was 2.9, incidence of successful multiple broods was 6%, annual productivity was 1.07 fledglings per female, and the breeding season lasted 22 weeks. These values are consistent with the basic life history traits of the *Malurus* genus and characteristic of Australian passerines.

Purple-crowned fairy-wrens occurred in small family groups in well defined territories that were often maintained throughout the year and often for successive years. As with previous studies (Emlen, 1991; Rowley and Russell, 1997; Brooker and Brooker, 2001) PCFW family groups in the VRD consist of a breeding pair and often included one or more helpers, most likely retained progeny. Most helpers were males. Female offspring probably dispersed earlier than males, a notion supported by research on other *Malurus* species (Rowley and Russell, 1997). I did not record any male helpers inheriting their natal territory to support the “benefits of philopatry” hypothesis (Stacey and Ligon, 1991). However, this could be a function of high senior male survival and only three years of observations.

The PCFW showed a strong affinity for *Chionachne* and this species plays a crucial role in all aspects of PCFW ecology (i.e. foraging, breeding and cover). Wrens were rarely found outside *Chionachne* habitat and when they were, only rarely would they stray more than 30 meters beyond the nearest *Chionachne* patch.

*Xanthium* is a highly invasive weed easily spread by burrs (Smith, 1995) and was ubiquitous at all sites, often forming dense thickets along the riparian corridor and especially along drainage lines and erosion channels (i.e. areas with abundant bare ground). While it could be assumed that *Xanthium* is a favorable component within the PCFW habitat mosaic because it would provide foraging and cover opportunities, this species is suitable only for a short period as it quickly dries, leaving little cover or food. Moreover, although birds will forage in *Xanthium*, it was not a preferred substrate. Because this weed may be a competitor of the preferred grass, *Chionachne*, *Xanthium* should be regarded as a negative component within PCFW habitat.

Although the PCFW has been known to breed throughout the year when conditions are suitable (Rowley and Russell, 1997), the results from my study indicate a clear breeding season that commences at the beginning of the dry season and ends when conditions are at their driest in September. The initiation of breeding appears to coincide with the recession of flood waters and subsequent recovery of *Chionachne* once waters recede. During the three years of my study, the Victoria River flooded and the majority of *Chionachne* habitat was inundated to some extent. It is possible that breeding may occur at different times of the years when the river does not flood and *Chionachne* is suitable for breeding.

Nest predation rates were found to be at least 51% but could be as high as 67%. Nest predation rates for other *Malurus* species have been reported to be as high as 55% (Mulder, 1992) however predation rates may vary greatly among and within species, depending on

climatic and habitat conditions (Brooker and Brooker, 2001). Nevertheless, the predation rate found during my study is one of the highest recorded for *Malurus* species (Rowley and Russell, 1997). Aside from two predation events attributed to cats and one to an avian predator, nest predators were not identified but could potentially include a range of species commonly seen at all sites. However, native rats (e.g. *Rattus villosissimus*) and exotic rats (*Rattus rattus*) were found at study sites and could potentially be a primary predator of wren nests. Clutch predation by rats characteristically leaves nests intact, such as were found during my study, and elsewhere rats have been shown to be significant predators of avian clutches (Major, 1991; Laurance and Grant, 1994).

Despite high nest predation rates, females managed to fledge an average of 1.07 young per year (i.e. per group year). Undoubtedly, this would not be possible without the large number of replacement clutches, frequently four that females laid in a season. Although it is true that during my study, females re-nested up to three times after a failed attempt, it is also very possible that during less favorable breeding seasons this may not be possible. Thus the likelihood of females producing young with such high predation rates would be very small.

The high levels of predation found during my study are of particular concern and need further investigation. In particular, comparisons between sites and identification of nest predators are crucial.

The overall combined survival rate for all sites was 61%, although survival rates were higher for 2001 through 2002 than 2002 through 2003. Overall adult survivorship is similar to other *Malurus* species (Rowley and Russell, 1997), although the survival rates at two of the sites within Gregory National Park are remarkably high, potentially linked to very favorable site conditions in combination with high rainfall.

The high annual adult survival rates, extended breeding season, and multiple broods found during my study suggest the PCFW is able to exploit favorable conditions when they arise and could produce a high recruitment rate. However, variable adult survival rates found between sites indicate that even in favorable years, adult survival can be compromised. Further, my study took place during three high rainfall years and may represent the “best case scenario” and may not reflect long term trends of the population in the VRD.

Details of differences (i.e. annual adult survival, group size, and productivity) among sites are dealt with in detail in Chapter 4.

### **Comparisons between Victoria and Drysdale River Populations**

There are clear structural differences between the riparian grass, *Chionachne*, inhabited by the PCFW in my study and the woody palm-like shrub, *Pandanus*, inhabited by the PCFW studied by Rowley and Russell (1993). Consequently, the data reveal considerable differences between the VRD population and the previously studied Drysdale River population. Some of these differences can be attributed to habitat utilization and are not surprising, if we consider the structural dissimilarity between the two habitat types (e.g. nest height, territory shape and size). Although not directly addressed in my study, preliminary differences in median body size between the two PCFW populations indicate further research is needed to determine the importance of these differences in regard to subspecies classification.

At Drysdale River, in *Pandanus* habitat, group size of PCFWs was smaller (2.28; Rowley and Russell, 1993) in comparison to groups along the Victoria River (2.61) ( $D = 0.3020$ ,  $P < 0.001$ ). At Drysdale River, 80% of groups consisted of a sole breeding pair (Rowley and Russell, 1993) whereas along the Victoria River 50% of groups consisted of a breeding pair and an additional 41% had one helper.

The shape of territories of the PCFW between *Pandanus* and *Chionachne* habitat also differed according to the distribution of each species. Because *Pandanus* exists in narrow strips along the banks of the river, but *Chionachne* forms dense stands that can vary greatly in shape and width, PCFWs at Drysdale River were rarely found more than 10 meters from the river, whereas, if *Chionachne* was present, they could be found more than a kilometer from the river in the VRD. *Chionachne* in some areas (e.g. GNP2, Fig. 3-6) was wide enough that it could support two wren territories extending back from the river. Hence, given a length of river, *Chionachne* habitat can potentially support more wrens than *Pandanus* habitat. However, this advantage may only be significant in undisturbed areas as *Chionachne* is also more prone to disturbances (e.g. grazing) than *Pandanus* (Officer, 1964; Boekel, 1979).

The prey items consumed by PCFWs reported by Rowley and Russell (1993) differ from the most common prey items consumed during my study. In my study, Lepidopterans (butterflies and moths) and Orthopterans (grasshoppers, crickets etc.) were the most common observed prey items taken by wrens, whereas Rowley and Russell (1993) reported a large number of Coleopterans (beetles), Hymenopterans (ants) and Hemipterans (bugs) as prey items. However, these differences are probably largely because of different methodologies. I used observational data that would be biased towards records of large conspicuous prey items in comparison to the fecal analyses used by Rowley and Russell that could in turn be biased toward prey with hard exoskeletons (1993).

Breeding data from the Drysdale River study indicated a bimodal breeding pattern, similar to the pattern found in some other tropical passerines and potentially related to food availability (Rowley and Russell, 1993; Noske and Franklin, 1999). Although there is a possibility that the difference in breeding pattern between the VRD and Drysdale River is due to site variables

(habitat or climate), Rowley and Russell (1993) suspected that the bimodal breeding pattern might simply reflect survey effort during the breeding season. Other *Malurus* species studied in depth (e.g. *M. splendens* and *M. pulcherrimus*) show no signs of a bimodal breeding pattern although neither one of these species are tropical in distribution (Rowley *et al.*, 1991; Brooker and Brooker, 2001). However, data from my study also indicate that a bimodal breeding pattern is also unlikely for the PCFW, at least in *Chionachne* dominated habitat.

Consistent with primary habitat use, PCFWs at Drysdale River primarily used *Pandanus* as a nesting substrate (96%) and *Chionachne* or *Mnesithea* (99%) in the VRD. Consequently, nests in the VRD were lower (307 mm) than those at Drysdale River (880 mm) consistent with nest substrate structure and height. Although nest predation rates at Drysdale River were unknown, higher nest location and placement in *Pandanus* (the serrated foliage of which may offer some additional protection) could have resulted in lower predation rates in *Pandanus* habitat.

Despite the high predation rates encountered in *Chionachne* habitat in the VRD, average productivity was higher in this habitat (1.07) in comparison to the *Pandanus* habitat (0.78) at Drysdale River (Rowley and Russell, 1993). There was high degree of variation in productivity between years at Drysdale River ranging from 0.26 (young per group), following a year with low rainfall to as high as 1.22 (young per group) following a year with high rainfall. There was little variation between years in the VRD (1.05, 1.09 and 1.06 young per group respectively from 2001 through 2003), however all three breeding seasons were preceded by high rainfall. If rainfall influences productivity in the VRD as much as it did at Drysdale River, productivity could potentially be extremely low in the VRD in low rainfall years if predation rates as high as they were in my study.

Annual adult survival was high in both *Pandanus* (70.7%) and *Chionachne* (60.7%) habitat, varying between years in both habitats, and in the VRD between sites (Chapter 4). Both habitat types can provide good cover against predators and predator evasion has been recorded in both habitat types. For example, in my study PCFW evaded predators on numerous occasions by diving down into the dense *Chionachne* vegetation. During flooding, however, *Pandanus* habitat would be less likely to be completely inundated and thus, be more resistant to destruction compared to *Chionachne*. Thus, during and following flooding events, other emergent vegetation may be important for the survival of wrens in *Chionachne* habitat.

The differences found in my study in comparison to previous research highlight the need for a broad range of site-specific studies in order to understand the biology and ecology of a species, especially when the habitat a species occupies and external influences within that habitat could differ significantly.

Table 3-1. Number of PCFW captures by year (2001-2003), excluding preliminary captures in 2000.

Individuals	2001	2002	2003
Female	19	25	10
Male	24	28	14
Juvenile	8	4	13
1 <sup>st</sup> capture/banded	51	57	37
Recaptures	1	20	11
Total Birds Captured	52	77	48

Table 3-2. Morphological measurements of captured individuals captured from 2001-2003. Mean  $\pm$  SD (range).

Individuals	Weight (g)	Wing Length (mm)	Tail Length (mm)	Tarsus Length (mm)
Female (n = 54)	10.6 $\pm$ 0.7 (9-12.5)	51.7 $\pm$ 1.5 (47.5-55)	68.7 $\pm$ 6.5 (49.09-87.9)	20.2 $\pm$ 1.4 (17.5-23.5)
Male (n = 66)	11.1 $\pm$ 0.6 (9.75-12.5)	54 $\pm$ 1.7 (50-57)	71.7 $\pm$ 5.4 (58.3-83.3)	20.98 $\pm$ 1.1 (18.27-23.26)
Juvenile (n = 26)	10.3 $\pm$ 0.6 (9.5-11.5)	51.6 $\pm$ 1.3 (49-54)	75.2 $\pm$ 4.7 (62.69-85.9)	19.9 $\pm$ 0.7 (18.51-21.27)

Table 3-3. Summary table of comparisons in differences between male, female and juvenile morphological measurements. F = female (n = 54), M = male (n = 66), J = juvenile (n = 25), W = weight, WL = wing length, TL = tail length, and TA = tarsus length (t-tests, significant differences identified at P < 0.05 by \*, P < 0.001 by \*\*, not significant by NS).

	Weight (g)			Wing length (mm)			Tail length (mm)			Tarsus length (mm)		
	F	M	J	F	M	J	F	M	J	F	M	J
F	-	**	*	-	**	NS	-	**	**	-	**	NS
M	**	-	**	**	-	**	**	-	**	**	-	**
J	*	**	-	NS	**	-	**	**	-	NS	**	-

Table 3-4. Group size and composition (n = 88 group-years).

Group Size and composition	2001 (n = 22)	2002 (n = 34)	2003 (n = 32)	Total no. (n = 88)	Total group sizes
2 (1M, 1F)	9	16	19	44	44
3 (2M, 1F)	12	10	12	34	36
3 (1M, 2F)	0	2	0	2	
4 (3M, 1F)	1	2	0	3	6
4 (2M, 2F)	0	3	0	3	
5 (4M, 1F)	0	1	1	2	2

Table 3-5. Nest location and measurements 2001-2003.

Nest substrate (n = 65)	<i>Chionachne</i> 85%, <i>Mnesithea</i> 14%, <i>Barringtonia</i> 1%
Nest entrance (n = 33)	Mean 40.5 cm (SD $\pm$ 12.2)
Nest base (n = 35)	Mean 30.7 cm (SD $\pm$ 14.2)
Nest entrance orientation (n = 33)	East 73%, West 12%, North 9%, South 6%
% Vegetation cover (n = 36)	Mean 65 % (SD = 15.1)

Table 3-6. Fate of nests with eggs (n = 51).

Fate of clutches	Number of nests (n = 51)
Fledged at least 1 young	12
Abandoned (eggs un-hatched)	5
Failed at egg stage (predation)	20
Failed at nestling stage (predation)	5
Unknown (but fledglings not located)	9



Figure 3-1. Adult male PCFW, full breeding plumage.



Figure 3-2. Adult female PCFW.



Figure 3-3. Eclipse male PCFW.



Figure 3-4. Juvenile PCFW.

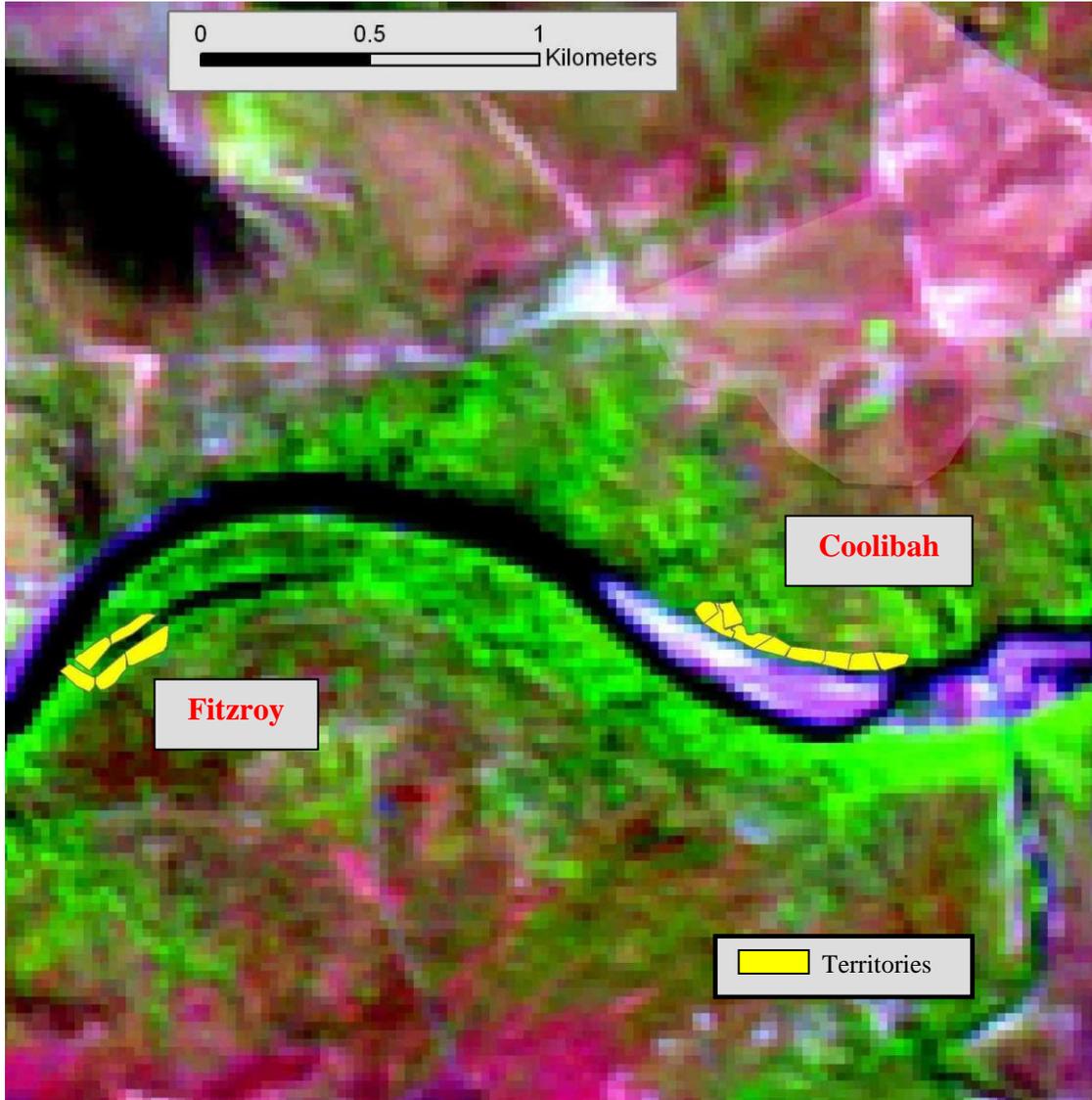


Figure 3-5. Location and shape of territories at Coolibah Crocodile Farm and Fitzroy Station, 2002.

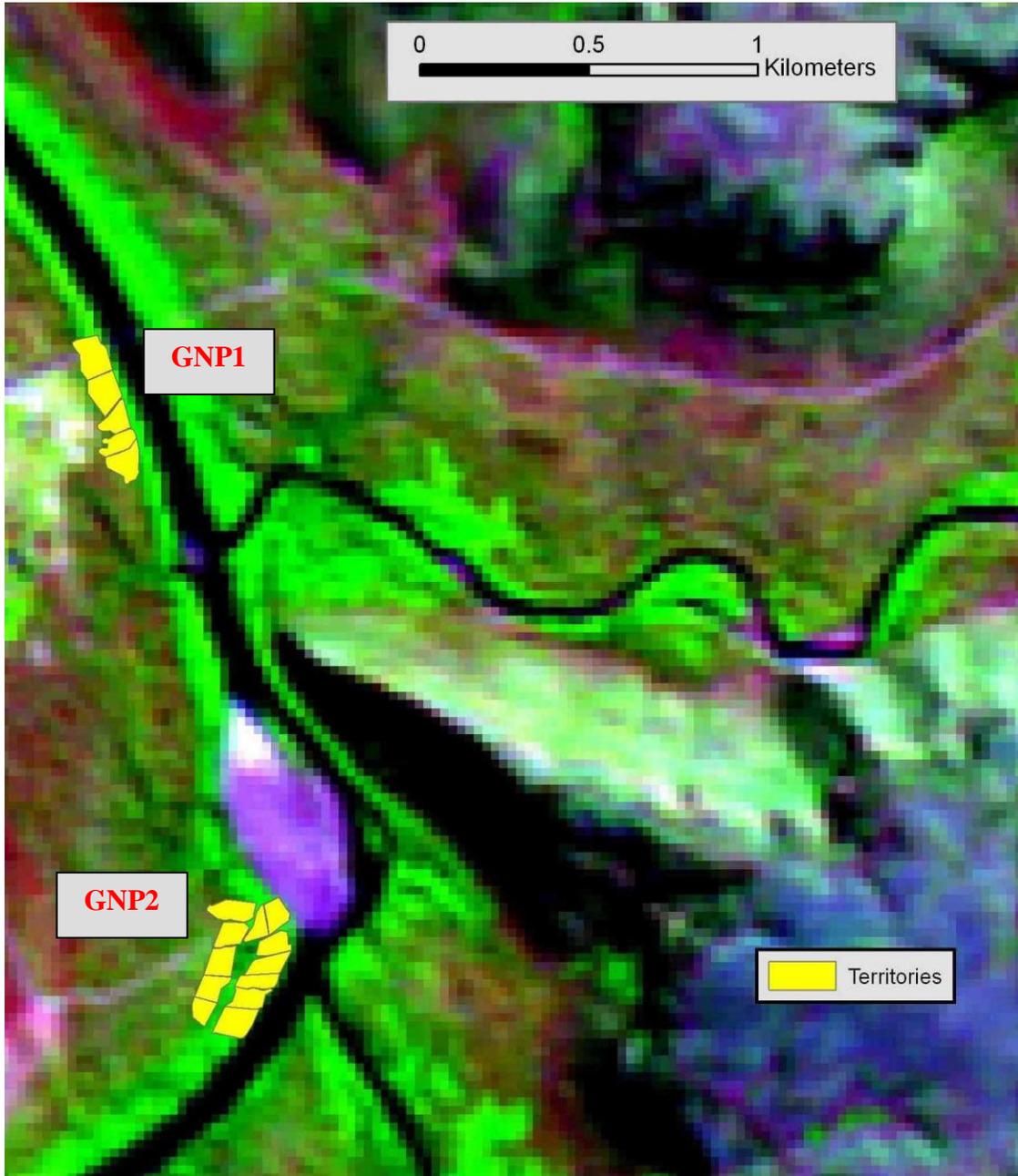


Figure 3-6. Location and shape of territories within Gregory National Park (GNP1 and GNP2).

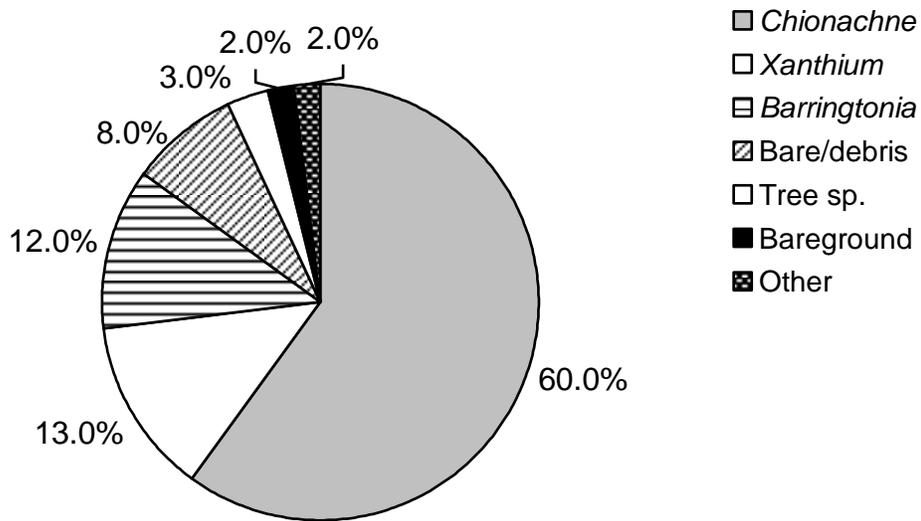


Figure 3-7. Time spent foraging (%) in various substrates (n = 963 min). Approximate height of primary foraging substrates; *Chionachne* (canegrass) = 1-3m, *Xanthium* (noogoora burr) = < 1m, *Barringtonia* (freshwater mangrove) = 4-8 m.

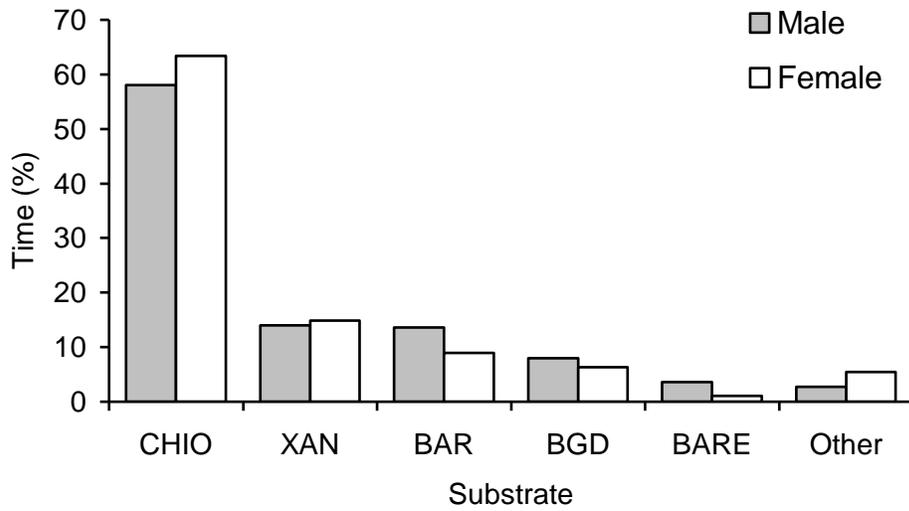


Figure 3-8. Male and female foraging preferences. Percent of time spent in each substrate. CHIO = *Chionachne*, XAN = *Xanthium*, BAR = *Barringtonia*, BGD = Bareground w/debris, BARE = Bareground (n = 900 min.).

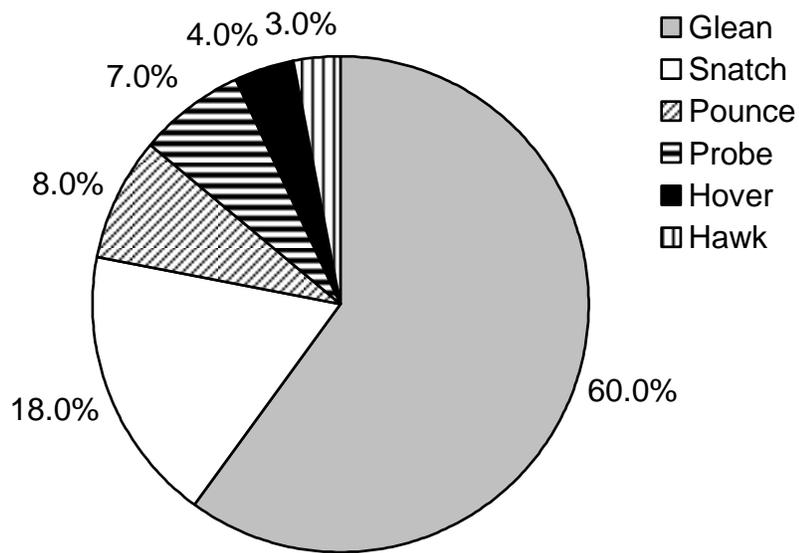


Figure 3-9. Foraging behavior. Number of times each behavior was observed, expressed in %. (n = 110).

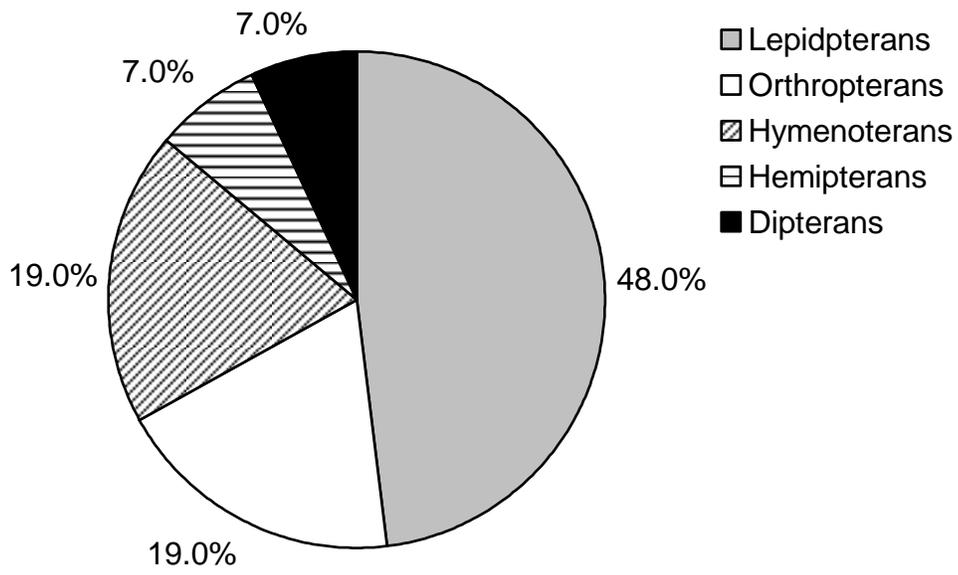


Figure 3-10. Prey items captured. Percent of total prey types, by insect order (n = 27).

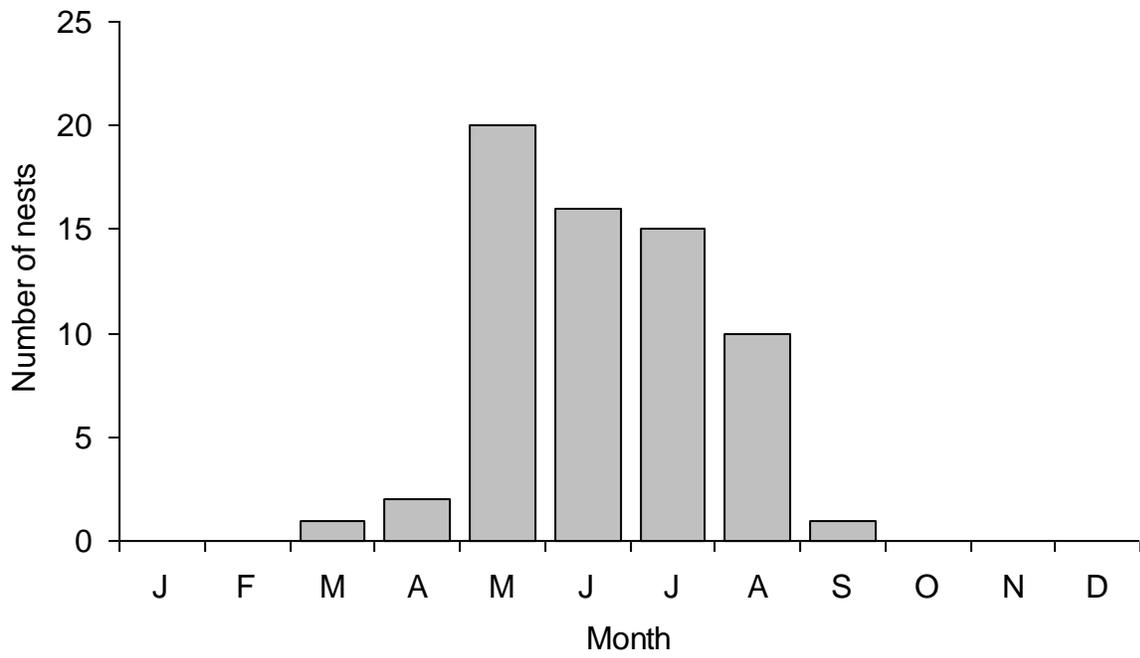


Figure 3-11. Number of nests found per month, 2001-2003 (n = 65).

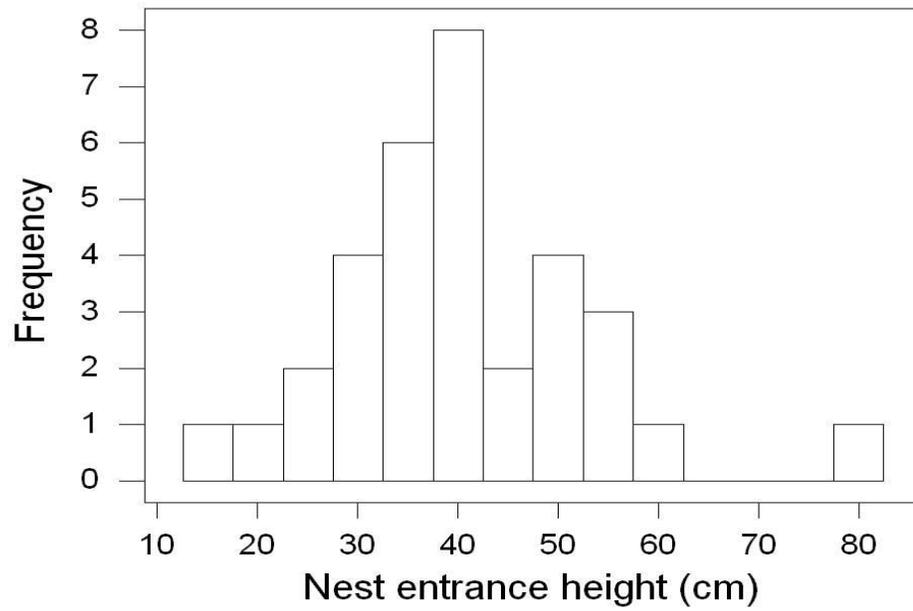


Figure 3-12. Nest entrance height (n = 33).

## CHAPTER 4

### RESPONSE OF THE PURPLE-CROWNED FAIRY-WREN TO DISTURBANCE FACTORS

The aim of this chapter is to investigate the primary threats to the PCFW population in the VRD. Specifically, I focus on the effects of cattle grazing as this disturbance has been implicated as a cause of recorded structural changes in riparian and other vegetation, and also been correlated with changes in avian diversity and abundance. The VRD presents an excellent opportunity for comparative measures as grazing was present in riparian areas at varying degrees. Weeds and fire are additional potential threats to the PCFW; however these are only discussed superficially in comparison to grazing for compelling reasons. Weeds occurred at all sites, making comparative analyses more difficult. Fire was witnessed only on a single occasion at a very small scale and is discussed in this context.

#### **Introduction**

In the southern regions of Australia, there has been a significant decline in avian diversity, and in the extent and quality of native vegetation. In some regions, habitat modification can be as high as 90%, and this loss has been implicated as one of the driving forces of bird decline (Recher and Lim, 1990; Ford *et al.*, 2001). Although the changes in faunal composition may not be as obvious as they are in the southern region, recent studies in northern Australia are demonstrating large scale yet subtle changes to avian populations as well (Franklin, 1999; Garnett and Crowley, 2000; Woinarski and Catterall, 2004; Hobbs, 2005). Although species responses vary, pastoralism and changing fire regimes may be the underlying causes of these changes (Woinarski, 1990; Crowley and Garnett, 1999; Woinarski and Ash, 2002).

The impacts of some disturbances, most notably fire, has been studied for several *Malurus* species (e.g. *Malurus elegans* and *Malurus splendens*) (Brooker and Brooker, 1994; Russell and Rowley, 1998). Habitat fragmentation has also been studied in detail for species

such as *Malurus pulcherrimus* (Brooker and Brooker, 2001) and the effects of fragmentation and fire have been modelled for the splendid fairy-wren, *Malurus splendens* (Brooker and Brooker, 1994). The Purple-crowned fairy-wren, however, has not been studied in this context even though it is confined to riparian habitat prone to disturbances such as weed invasions and grazing. Anecdotal evidence has suggested that grazing, erosion, and weeds are the primary causes of PCFW population declines and fragmentation (Smith and Johnstone, 1977; Garnett, 1992; Rowley, 1993) but until the current study, there has been no concrete evidence of such effects.

Elsewhere, the effects of grazing on bird species in general have been the subject of numerous studies that have overwhelmingly concluded that grazing has a negative influence on avian diversity and nest success (Croonquist and Brooks, 1993; Ammon and Stacey, 1997; Krueper *et al.*, 2003; Scott *et al.*, 2003). Grazing in particular changes understory composition and structure (Fleischner, 1994; Jansen and Robertson, 2001; Martin and Possingham, 2005; Werner, 2005), these changes may be one of the most important effects of grazing on the PCFW as it is dependent on the understory for foraging, breeding and cover. For example, Martin and Possingham (2005) found that foraging substrate was a good indicator of how an avian species would cope under grazing pressure. Thus, species foraging at lower and medium levels of the vegetation strata would be influenced by grazing at higher levels than species that forage in the upper canopy. In the case of the PCFW, the results from Chapter 3 reveal that the majority of foraging takes place at lower levels of the vegetation with little time spent in the middle to upper vegetation levels.

Riparian zones tend to be exposed to higher rates of disturbances than other habitats and riparian obligates such as the PCFW have evolved with endogenous disturbances such as annual flooding and occasional fires. However, exogenous disturbances within the riparian zone, such as grazing by domestic stock, are relatively new and as a species that spends the

majority of its time within four meters of the ground, the PCFW may be particularly susceptible to changes (structurally and floristically) in the understory.

In the VRD, 59% of the land is used for pastoralism, and supports 545,000 cattle (Karfs and Trueman, 2005). Aside from the economic benefits of pastoralism, the industry carries strong historical ties to an era whose sentiment is still felt strongly throughout Australia. However, the industry has contributed to the modification of savannas in northern Australia and this modification has influenced faunal composition (Hobbs, 2005).

I used a comparative approach to determine the threats to the PCFW and its primary habitat focusing on grazing effects, using site-specific habitat variables and site-specific ecological and life-history parameters (group size, adult survivorship, and productivity). I focused on comparisons between grazed and ungrazed sites in addition to pre and post grazing comparisons, to determine relative differences. The roles of fire and weeds are included superficially based on the limited data available for these variables, and the lack of scope in my study for manipulating these variables.

## **Methods**

### **Vegetation and Site Condition**

Grazing pressure by cattle varied between sites. Coolibah was heavily grazed (from 2002); Dashwood had a history of heavy grazing during my study but was now lightly grazed; Fitzroy was lightly/moderately grazed; and the two sites within Gregory National Park (GNP1 and GNP2) were ungrazed. These grazing categories were subjective, based primarily on the number of cattle observed at sites; nevertheless, they provided a baseline for comparative analyses (also see Table 2-3, Chapter 2).

At each of the five fixed study sites, I mapped vegetation using a point intercept method along 50-meter transects. Transects (two in each PCFW territory) were placed in a diagonal line, parallel to each other to ensure the maximum coverage possible and were

placed at least 20 meters apart (Fig. 4.1). At each one meter intersect (point data) I recorded the following; substrate (plant species, bareground, bareground w/ debris, or dead trunk/branch), substrate height, canopy cover (species) and height. In addition, the substrate located within 20 cm radius of the intercept (plot data) was also recorded. To assess wallaby and cattle densities I recorded the presence or absence of wallaby feces and cattle feces and hoof prints within a 20 cm radius of each one meter intercept. Unfortunately, it was not possible to conduct two transects for all PCFW territories due to the high prevalence of buffalo and scrub bulls which made it impossible to conduct surveys safely. On another occasion it was impossible to finish all transects at Coolibah due to the interference of an adjoining stakeholder.

All data were collected during the 2002 dry season (June-September) when all five sites had been established and PCFW territories had been identified. I chose this time for logistical reasons as three of the sites were not accessible during the wet season. Further, this period coincided with the PCFWs breeding season a time when vegetation cover and structure might be more important for nesting and foraging.

Differences among sites for each environmental variable were analyzed using ANOVA. Tests were followed by a-posteriori Tukey's LSD mean comparison procedures when variables were significant in the ANOVA at the  $P \leq 0.05$  level. Percentage values were arcsine transformed prior to statistical analyses (Fowler *et al.*, 1998).

Overall habitat differences among sites were analysed using ordination (multi-dimensional scaling) to portray variation across all measured habitat variables described above (except for the two variable cattle feces and wallaby feces). This was undertaken in the program PRIMER (Clarke and Gorley, 2001), and was based on the Klyczynski similarity index. A two-dimensional solution is presented.

The extent to which the matrix of pair-wise similarity values was associated with site was examined using ANOSIM (Clarke and Green, 1988).

## **Fire**

Over the course of my study there was only one incidence of fire, and the extent was minimal with only three territories affected to any great extent. Although I witnessed the fire, it occurred before vegetation data was obtained. Photographic evidence does show the regeneration of *Chionachne* at this site and it was also possible to document PCFW mortality and re-colonization post-fire.

## **Variation Among Sites**

The methods I used to collect ecological data on the PCFW are detailed in Chapter 3. Differences in PCFW biology and ecology among sites were analyzed using all sites unless sample sizes were too small, these are indicated where present. At GNP1 and GNP2 (both ungrazed), all sites and transects were used. At Coolibah pre-grazing and post-grazing measures were possible for a number of PCFW variables (i.e. group size, adult survival and productivity). Sample sizes at Fitzroy for PCFW biological and ecological data were too small for comparative analyses. Dashwood had been continuously grazed during all my field seasons making any comparisons between years superfluous.

Foraging preferences of PCFW at one site (GNP2) served as the base-line data to examine this variable. I did not want to collect data at grazed sites to avoid any potential confounding effects of grazing. Data to determine foraging preferences were based on number of minutes per foraging substrate (see Chapter 3 for a full description of methods for foraging data) and substrate abundances were based on plot data based on 17 transects conducted at this site (GNP2). A description of vegetation transects was described previously in this chapter.

Comparisons among sites of PCFW ecological variables (i.e. annual adult survival, group size, productivity) were tested using either a Mann-Whitney U test (for comparisons between two groups/sites) or a Kruskal-Wallis test (for comparisons involving more than two sites): I used these non-parametric tests as the assumption of normality could not be made for these variables. I used a Chi-square test to compare the frequency distribution of foraging sites (substrates) against the actual proportion of those sites in the environment (using Minitab 13.1, Minitab Inc., Coventry, UK).

## Results

### Vegetation and Site Condition

The incidence of grazed *Chionachne* was highly variable between sites; two sites (GNP1 and GNP2) displayed no evidence of grazed *Chionachne*, two sites had similar amounts of grazed *Chionachne* (Dashwood; 31% and Fitzroy; 39%) and at Coolibah the majority (93%) of *Chionachne* was grazed (Fig. 4-2,  $F_{4,70} = 135.2$ ,  $P < 0.001$ ).

Cow and wallaby density varied among sites (Fig. 4-3). Cow evidence (feces and hoof print) was the most prevalent at Coolibah and was higher than any of the other sites ( $F_{4,70} = 31.7$ ,  $P < 0.001$ ). Cow evidence was similar for Dashwood and Fitzroy and non-existent at GNP1 and GNP2. Wallaby evidence (feces) also varied between sites and was highest at Dashwood (40%) followed by Coolibah (32%) and Fitzroy (20%). Wallaby evidence was much lower at GNP1 (2%) and GNP2 (0.2%) than any of the other sites ( $F_{4,70} = 92.8$ ,  $P < 0.001$ ).

Figures 4-4 through 4-6 illustrate the change in *Chionachne* cover that occurred when cattle were moved onto the site at Coolibah in May, 2002. Figure 4-4 illustrates the site just before cattle arrived and Figs 4-5 and 4-6 after cattle had been at the site for approximately 4 months. Although the base of the *Chionachne* is still present, the change in cover is clearly

visible. Note that the timing of cattle arriving, at the end of May, coincided with the peak of the breeding season.

I sampled seventy five vegetation transects in 2002 representing 39 territories at the five fixed field sites. Twenty-two transects (14 PCFW territories) were conducted at Coolibah, five (2.5 PCFW territories) at Dashwood, 14 (seven PCFW territories) at Fitzroy, and 17 (eight and a half PCFW territories) at both GNP1 and GNP2. There were only a few groups present at the Dashwood Crossing site accounting for the low number of transects at that site. Note that in the corresponding figures, Coolibah, Dashwood and Fitzroy represent the grazed sites and GNP1 and GNP2 represent the two ungrazed sites located within the national park.

Ground cover was not very diverse, dominated by one plant species, *Chionachne cyathopoda* (Fig. 4-7). Overall, *Chionachne* accounted for 19% of point data and 39% of plot data. For point data, *Chionachne* frequency ranged from 10% at Dashwood to 26% at GNP2 (Fig. 4-8). *Chionachne* frequency (plot data) was significantly lower at Coolibah than GNP2 but not GNP1 ( $F_{4,70} = 5.8$ ,  $P < 0.001$ ). *Chionachne* frequency of plot data was similar to that of point data, ranging from 21% at Dashwood to 49% at GNP2. Dashwood had the overall lowest coverage in comparison to all other sites ( $F_{4,70} = 6.4$ ,  $P < 0.001$ ).

*Chionachne* height, at height intervals of <1 m, 1-2 m, and >2 m, differed among sites (Fig. 4-9). *Chionachne* greater than two meters in height was most prevalent at the two ungrazed sites, GNP1 (40%) and GNP2 (46%), no *Chionachne* greater than two meters was recorded at either Coolibah or Dashwood. Although *Chionachne* more than two meters was less common at Fitzroy (32%), this difference was not significant. Coolibah had a disproportionately larger number of *Chionachne* stands less than one meter in height (70%) in comparison to all sites except Dashwood ( $F_{4,70} = 19.7$ ,  $P < 0.001$ ). In contrast, the two sites that were not exposed to any grazing by cattle (GNP1 and GNP2) had very few stands

that were less than 1 meter in height (14% and 8% respectively). Medium height (1-2 meters) *Chionachne* stands were less prevalent at Coolibah than the remaining sites ( $F_{4,70} = 2.8$ ,  $P = 0.031$ ).

Bareground accounted for an overall 22% of point data (Fig. 4-7). Bareground was most common in erosion gullies and along drainage lines where it was often extensive. I found a substantial difference in bareground incidence between sites, most notably at Coolibah, where bareground was prevalent at both at points (42%) and plots (33%). In contrast, GNP2 had a lower frequency of bareground; bareground accounted for only 8% of point data and 5% of plot data (Fig. 4-10). Relative to the other sites, Coolibah had a higher frequency bareground than Fitzroy, GNP1 and GNP2 but not less than Dashwood; further, GNP2 had a higher frequency of bareground than Dashwood and Fitzroy but not GNP1 (point data,  $F_{4,70} = 21.3$ ,  $P < 0.001$ ). Within plots, Coolibah had an even higher frequency of bareground in comparison to all other sites including Dashwood, and Fitzroy had a higher incidence of bareground than either of the ungrazed sites GNP1 or GNP2 (plot data,  $F_{4,70} = 38.7$ ,  $P < 0.001$ ).

Bareground with debris (e.g. leaves and dried *Chionachne* blades) was the dominant non-vegetative cover and accounted for 53% of point data and 26% of plot data (Fig. 4-7). Debris was often very thick, especially at sites where *Chionachne* was very dense when a thick layer of dried blades was often present (e.g. GNP2). The amount of bareground with debris varied between sites with the lowest frequency found at Coolibah (41%) and the highest at GNP2 (59%) (Fig. 4-11, point data  $F_{4,70} = 5.8$ ,  $P < 0.001$ ). However, the difference between sites using the plot data was less distinct although Fitzroy had more bareground with debris than Coolibah and GNP1 (plot data  $F_{4,70} = 3.6$ ,  $P = 0.01$ ).

The invasive weed *Xanthium* was the second most common understory plant species encountered at study sites (overall frequency; 2% of point data, 9% of plot data, Fig. 4-17).

*Xanthium* frequency at points was greatest at Dashwood (6%) followed by GNP2 (5%) in comparison to the sites, Fitzroy and GNP1 which had the same amount of coverage (2%) and no *Xanthium* was present at Coolibah (Fig. 4-12a, point data,  $F_{4,70} = 12.6$ ,  $P < 0.001$ ). Despite relative low frequency of *Xanthium* at points, *Xanthium* was present at higher frequency within plots, although the pattern of frequencies between sites remained constant. More *Xanthium* was found at Dashwood (26%) than either Coolibah (2%), Fitzroy (8%) and GNP1 (6%) but not significantly more than at GNP2 (18%) which had more *Xanthium* than Coolibah and GNP1 (plot data,  $F_{4,70} = 9.7$ ,  $P < 0.001$ ). Figure 4-12b illustrates the extent of *Xanthium* along the river bank at Dashwood.

In addition to *Xanthium*, two other weed species, *Passiflora foetida* and *Ricinus communis*, were also found at some of the study sites. The frequency of the three combined weeds (*Xanthium*, *Passiflora*, *Ricinus*) followed a similar pattern to the distribution of *Xanthium* alone (Fig. 4-13). Dashwood and GNP2 had the highest incidence of weeds both points (6% at each site) and within plots (26% and 22%). Although Coolibah had the lowest incidence of weeds at both points (0.3%) and within plots (3%) in comparison to all sites, the difference was significantly less than at Dashwood and at GNP1, but not less than at Fitzroy (2%, 8%) and GNP1 (2%, 7%) which had similar weed frequencies (point data,  $F_{4,70} = 10.0$ ,  $P < 0.001$ ; plot data,  $F_{4,70} = 11.6$ ,  $P < 0.001$ ).

Total canopy cover varied among sites ( $F_{4,70} = 13.8$ ,  $P < 0.001$ ), with the highest cover found at GNP1 (72%), GNP2 (67%) and Dashwood (65%) (Fig. 4-14). Coolibah and Fitzroy had statistically significantly lower canopy covers, 24% and 38% respectively, in comparison to the national park sites GNP1 and GNP2, however, Fitzroy did not differ significantly statistically from Dashwood.

Four species comprised the canopy cover. These included two Eucalypt species, *E. microtheca* (coolibah or flooded Box) and *E. camaldulensis* (river red gum), *Barringtonia*

*acutangula* (freshwater mangrove) and *Ficus coronulata* (river fig). *E. microtheca* (58%) was the most common canopy species followed by *E. camaldulensis* (27%), *Barringtonia* (6%) and *Ficus* (4%) (Fig. 4-15).

Canopy species differed among sites, primarily in the relative abundances of *E. microtheca* and *E. camaldulensis* (*E. microtheca*  $F_{4,70} = 4.5$ ,  $P = 0.003$ ; *E. camaldulensis*  $F_{4,70} = 4.7$ ,  $P = 0.002$ , Fig. 4-16). *Eucalyptus microtheca* was the dominant canopy cover at all sites with the exception of GNP2 where *E. camaldulensis* was more abundant. *E. microtheca* was especially dominant at Dashwood, where *E. camaldulensis* was completely absent. *Barringtonia* was more common at GNP1 than elsewhere but this difference was significantly larger only than that at Coolibah ( $F_{4,70} = 3.9$ ,  $P = 0.006$ ). *Ficus* was present in low densities at all but one site (Dashwood) and was most common at GNP2 but abundance differences among sites did not vary greatly ( $F_{4,70} = 0.52$ ,  $P = 0.719$ ).

Overall, there was substantial variation in PCFW habitat across the set of five sites, as determined by an ordination of all transects by their habitat variables (Figure 4-17). This ordination has an acceptable stress value of 0.20. The variation is most marked between the two national park sites (GNP1, GNP2) on one hand (with no overlap of groups of clusters), and the heavily-grazed site (Coolibah) on the other, the less intensively grazed sites (Dashwood and Fitzroy) fall between these clusters.

This distinction among sites was also corroborated by ANOSIM, where the Global R statistic was 0.60 ( $p < 0.001$ ), indicating that there was a substantial overall variation in environmental variables among sites. Pair-wise comparisons among sites are given in Table 4-1; these show significant environmental variation between all pair of sites (other than between Dashwood (where sample size was small) and Fitzroy): differences were most marked between Coolibah and each of the two national park sites (GNP1 and GNP2).

## **Fire**

An area of GNP2 was burned in October 2001 in an effort to create a fire break from an approaching fire. As it crossed the study site, the fire was initially very hot (consuming some of the canopy) but soon (within 50 meters) slowed down and burned only patches of ground-level vegetation. The understory of three PCFW territories was burnt severely (Fig. 4-18a). During the fire one adult female disappeared; she was never seen again and was presumed to have perished. The male of the original pair was subsequently located two territories from his original territory. Regeneration of *Chionachne* occurred rapidly and three months later (December 5, 2002), a reasonable understory was present (Fig. 4-18b) and birds were found foraging in the area. By February, 2002, this territory was once again occupied by a breeding pair. During that year's breeding season, all three territories that had been heavily burned in 2001 were once again occupied by breeding pairs.

## **Variation Among Sites**

The number of PCFW captured and banded at the five field sites is summarized in Table 4-2.

Average group size among sites for combined years (2001-2003) varied significantly between sites (Fig. 4-19) ( $H = 2.0$ ,  $P = 0.058$ ). Fitzroy was not included in the analyses as the sample size was very small ( $n = 4$ ). Numerically, the lowest average group size was found at Dashwood (2.1) and GNP2 had the highest (2.8). Group size was the same at Coolibah and GNP1 (2.7) and slightly lower at Fitzroy (2.4). Comparisons of group sizes at Coolibah pre- and post- grazing indicated a significant reduction in group size when cattle entered the site; average group size pre-grazing was 3.0 in contrast to post-grazing where mean group size was 2.2 ( $U = 141.5$ ,  $P = 0.014$ ).

I found substantial differences in adult survivorship among sites ( $H = 13.7$ ,  $P = 0.003$ , Fig. 4-20). Adult survivorship for two combined years (2001-2002 and 2002-2003) was

highest within the national park, 96% at GNP1 and 83% at GNP2. Coolibah and Dashwood had lower survival rates, 51% and 30% respectively. Survival rate was lowest at Fitzroy (29%) however this was based on only one year (2001 to 2002) and as this site was accessed infrequently it was not included in the statistical analysis.

Survival rate at Coolibah was much higher from 2001 through 2002 (90%) before cattle arrived in comparison to 2002 through 2003 after cattle arrived (26%), a 64% loss of adult banded birds. The median difference between these two years was statistically significant ( $U = 53.5, P < 0.01$ ).

Purple-crowned fairy-wrens showed a preference for particular foraging substrates as time spent foraging in particular substrates was not proportional to the amount of substrate present (Fig. 4-21) ( $\chi^2 = 15.282, P < 0.01$ ). Most notably, *Chionachne* and *Barringtonia* were preferred foraging substrates and bareground and bareground with debris were avoided, the difference between *Xanthium* abundance and foraging time was not significantly different.

I found some difference between group productivity among sites, the number of fledglings produced per breeding female was highest at Dashwood (1.36), Coolibah (1.29) and GNP2 (1.22), with lower rates were found at GNP1 (0.83) and Fitzroy (0) (Figure 4-22), however, this difference was not statistically significant ( $H = 1.94, P = 0.586$ ). Although Dashwood had the highest group productivity, this does not reflect the density of PCFWs at that site. Dashwood had the lowest mean group size and high adult mortality rates, thus the actual density of PCFWs at this site was very low in comparison to other sites. The unexpected low productivity rate at GNP1, within ungrazed Gregory National Park, may be attributed to high predation rates at this site. Of the 20 clutches found at GNP1, 10 (50%) were eaten, and three (15%) were abandoned; only four (20%) were successful and the fate of three (15%) was unknown but likely eaten.

## Discussion

I found some significant differences between sites in relation to grazing pressure and my results support the premise that grazing can have a profound effect on riparian vegetation structure and the PCFW population along the Victoria River. The differences found in the understory between grazed and ungrazed sites are not unexpected, especially at Coolibah where grazing pressure was intense, but the differences among sites was further bolstered by changes seen over time at a single site where birds were studied pre and post grazing. The reduction in group size and adult survivorship of the PCFW at the intensely grazed site occurred swiftly and were alarming.

That grazing results in significant changes in vegetation structure and that these changes in vegetation vary according to grazing intensity is, in itself, not particularly surprising. It is consistent with previous studies that have shown changes in vegetation structure as a result of grazing, and that high stocking rates cause the most damage to riparian vegetation (McIntyre and Lavorel, 1994; Yates *et al.*, 2000; Jansen and Robertson, 2001; Krueper *et al.*, 2003). During the dry season, cattle congregate along the river where they have easy access to water. *Chionachne* is a palatable grass and when other species of grasses become dry, *Chionachne* often remains green for a longer period, making it an attractive food source for cattle along the riparian corridor.

Coolibah, despite being heavily grazed, still retained stands of *Chionachne*, suggesting that one season of grazing is not enough to destroy the root base. In contrast, at Dashwood, a site grazed over a long period of time, there were fewer stands of *Chionachne* than any other sites (both grazed and ungrazed) implying that long-term grazing has an influence not only on *Chionachne* height but also on overall abundance. *Chionachne* height varied significantly among sites being much shorter at grazed sites, thereby reducing the amount of cover available for PCFWs.

At grazed sites, wallaby density was higher than at both ungrazed sites. Wallabies were also present at one of the national park sites (GNP1) where they congregate at the watered campsite adjacent to the river where green grass is always available. The thick reed-like properties of healthy *Chionachne* may not be attractive to wallabies, and, indeed I found no evidence of *Chionachne* being eaten by wallabies at GNP1. However, the regrowth after cattle-grazing, floods and fire is soft and tender, making it attractive to wallabies and during this time there may be substantial influence of wallaby grazing on *Chionachne* cover (Meers and Adams, 2003). Agile wallabies (*Macropus agilis*) are known to eat a diverse diet allowing them to exploit a vast array of resources in a variable environment (Stirrat, 2002), particularly in the dry season, so *Chionachne* regrowth may be a valuable resource to them. However, any potential grazing pressure by wallabies on *Chionachne* regrowth would most likely have little effect on the grass abundance compared to intense grazing by cattle; further their effects on soil would be less severe (Bennett, 1999). This notion is supported by research on pasture grazing by black-striped wallabies (*Macropus dorsalis*) where, despite a widespread belief in the significant effect of wallaby grazing on pasture, the greatest effect on vegetation structure was cattle grazing (Baxter *et al.*, 2001).

Weeds, in particular *Xanthium*, are pervasive through the riparian corridor of the Victoria River and *Xanthium* occurred at most of the study sites. At Coolibah, *Xanthium* was less abundant, because high stock densities caused an overall reduction in all herbaceous ground cover due to trampling, resulting in the large proportion of bareground encountered at this site (results comparable to studies elsewhere: Fleischner, 1994; Jansen and Robertson, 2001). *Xanthium* was most prevalent at Dashwood, however, one of the sites within the national park (GNP2) also had a very high incidence of *Xanthium*. This is perhaps a reflection of the large number of open areas (drainage lines and flood valleys) present at the national park site (GNP2) site. Although PCFWs foraged in *Xanthium*, this was not a

preferred foraging substrate compared to *Chionachne* and *Barringtonia*. Perhaps some cover is better than none (bareground). Importantly, my study indicated that PCFWs will exploit this resource if it is available, but it is equally important to consider that *Xanthium* is only suitable for foraging only for a short period of time (from the time it is tall enough to provide cover until it dries), and that this time does not extend through the breeding season. Furthermore, this species is not suitable for nesting, as the single stems could not support a nest, and may be negative component if it competes with *Chionachne*.

Difference in canopy cover may be attributable to flood dynamics as extensive flooding in this region is common and, based on water flow dynamics, floods are an important moderators of riparian vegetation (Start and Handasyde, 2002). During flooding events, canopy may become more important as *Chionachne* may be inundated. Opportunistic observations of PCFWs during flooding events found birds foraging in most canopy species found within the riparian zone. During flood events, which occurred during all three field season, PCFW were observed in *Barringtonia*, *Ficus* and *E. microtheca*, substrates not noted to be used frequently during the non-flood times that comprised the bulk of my fieldwork.

Although fire was only witnessed once during the course of my study, I did record some associated adult mortality in the PCFW territory that burned the hottest. *Chionachne* regenerated quickly after fire, with adequate rainfall, and was inhabited the following season. It is important to note that regeneration of *Chionachne* after fire was dependent on the timing of rainfall subsequent to the fire. One of the primary concerns about fire in *Chionachne* stems from the fact that it grows in long connected corridor usually without appropriate fire breaks, allowing fire to spread throughout a large area. Because *Chionachne* often exists in monospecific stands, a widespread fire would leave little cover for PCFWs in a large area until regeneration. Furthermore, any presence of grazing animals at a site following fire could further hamper regeneration of *Chionachne*, the PCFW's major habitat.

For a species whose individual annual survival rates can be over 90% (e.g. within Gregory National Park), the loss of 74% of adult birds in one breeding season (e.g. at Coolibah when cattle were introduced) is great cause for concern. No similar decrease in adult survival was recorded at either national park sites, and it is unlikely that some large-scale environmental factor was responsible for the change to PCFW recorded at Coolibah.

On the other hand, despite the decrease in adult survivorship and group size at the intensely grazed site (Coolibah), group productivity did not decrease. The remaining birds still produced an average of 1.09 juveniles per group (i.e. per breeding female), very close to the average annual productivity across all sites. The same was true at Dashwood, where despite relatively high adult mortality rates and smaller group sizes, group productivity rates were similar to those at ungrazed sites. The long-term perspective of the PCFWs at heavily grazed sites may not be as secure as at ungrazed sites, had I been able to determine juvenile survival. It is likely that juvenile survival at Coolibah after grazing began may have been very low, judging by the fact that very few birds were retained as helpers within their natal territory. It is important to note that despite high productivity at Dashwood and Coolibah the total number of birds produced at these sites was low, hence, these sites retain overall low PCFW densities in comparison to the other ungrazed sites.

At one of the national park sites (GNP1), group productivity was low, even though adult survival was high. This site had good habitat qualities; it was not subjected to grazing, *Chionachne* was abundant, and weed presence was lower than the other national park site (GNP2). This suggests that habitat quality may not be, at least directly, totally responsible for a low productivity rate. However, nest predation rates seemed to be very high at this site (Chapter 3), and perhaps this factor resulted in lower productivity rates than the second site within the national park (GNP2).

In summary, changes attributable to intense grazing (i.e. at Coolibah) resulted in a rapid reduction in *Chionachne* height and hence cover for the PCFW, which spends most of its time in lower vegetation (Chapter 3). Most importantly, an extremely high level of adult mortality marked the transition of one site from ungrazed to grazed within a single breeding season, strongly suggesting that grazing is a primary threat to PCFW populations in the VRD. Long-term grazing (i.e. at Dashwood) was associated with a reduction in *Chionachne* cover, an increase in weed abundance and high adult mortality of PCFWs.

Purple-crowned fairy-wren conservation management is the topic of the next chapter.

Table 4-1. Pair-wise differences between sites in the set of all environmental variables measured. Values in body of table give R, with associated probability levels marked as ns = not significant, \* p<0.05, \*\* p<0.01 and \*\*\* p<0.001.

Site	GNP1	GNP2	Coolibah	Fitzroy
GNP2	0.16 **	-	-	-
Coolibah	0.92 ***	0.92 ***	-	-
Fitzroy	0.27 ***	0.40 ***	0.65 ***	-
Dashwood	0.36 *	0.32 *	0.84 ***	0.001 ns

Table 4-2. Purple-crowned fairy-wren captures per site.

Year/Site	Coolibah (grazed)	Dashwood (grazed)	Fitzroy (grazed)	GNP1 (ungrazed)	GNP2 (ungrazed)
2001	12	2	8	4	7
2002	29	11	9	12	20
2003	24	6	0	9	9
Total no. captured	65	19	17	25	36

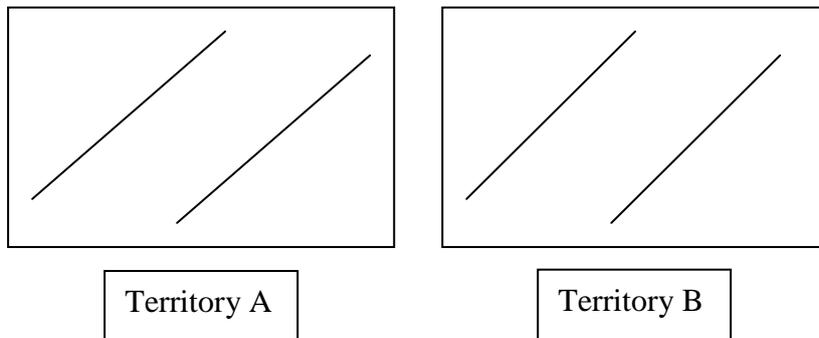


Figure 4-1. Positioning example of 50 meter transects within territories.

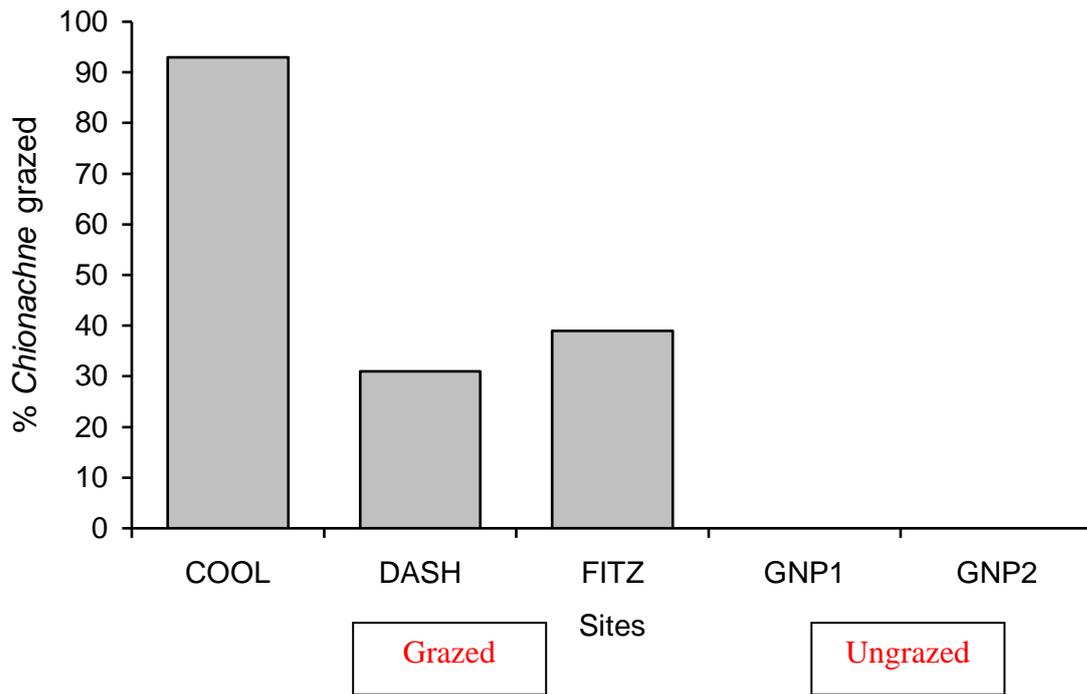


Figure 4-2. Frequency (%) of grazed *Chionachne* at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

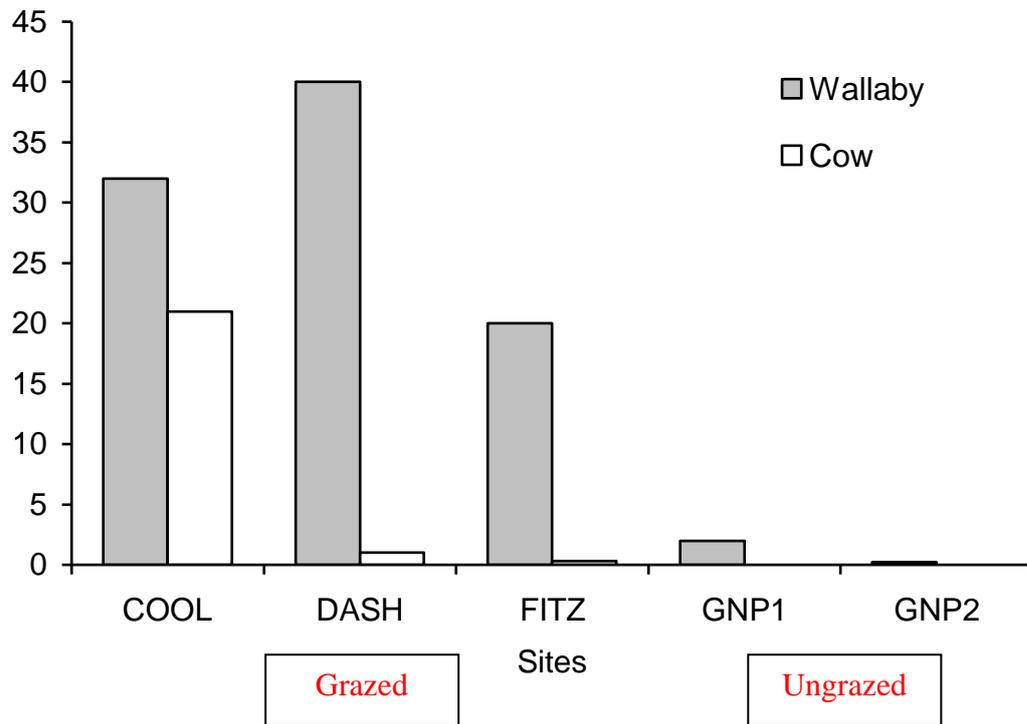


Figure 4-3. Wallaby (*Macropus agilis*) and cow (*Bos Indicus*) evidence at individual study sites. Wallaby evidence measured as % feces and Cow as % feces and hoof prints found (plot data) (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.



Figure 4-4. *Chionachne* (canegrass) stands in May 2002 at Coolibah before cattle were introduced.



Figure 4-5. *Chionachne* (canegrass) stands, October 2002, four months after cattle arrived at Coolibah, October 2002.



Figure 4-6. *Chionachne* (canegrass) stands after heavy grazing at Coolibah, October 2002.

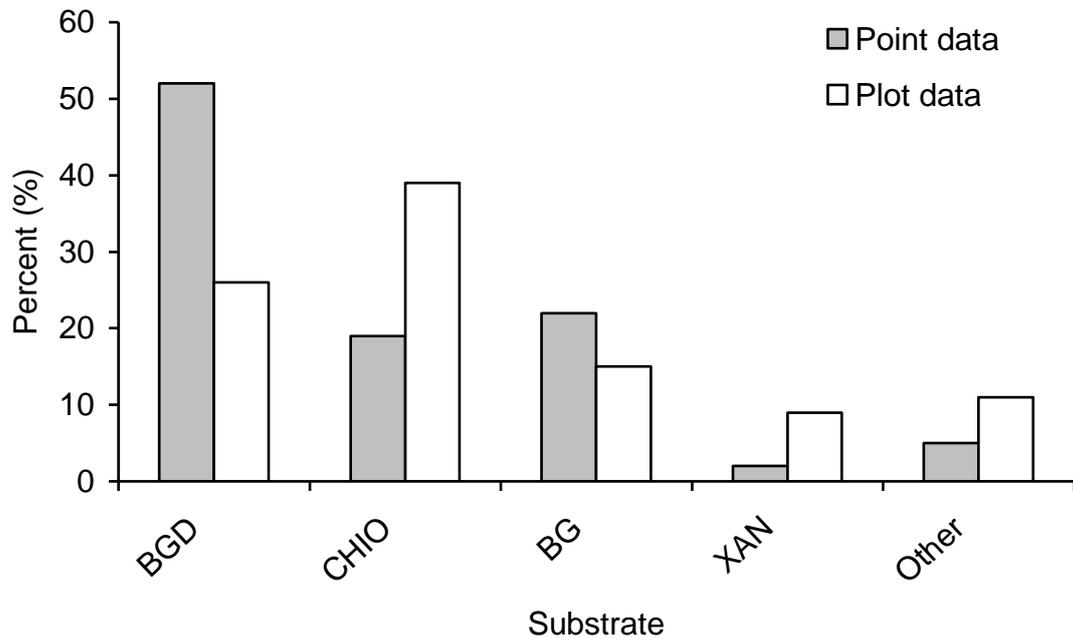


Figure 4-7. Groundcover frequencies (%) at study sites. BGD = Bareground w/ debris, CHIO = *Chionachne cyathopoda*, BG = bareground, and XAN = *Xanthium strumarium* (n = 75).

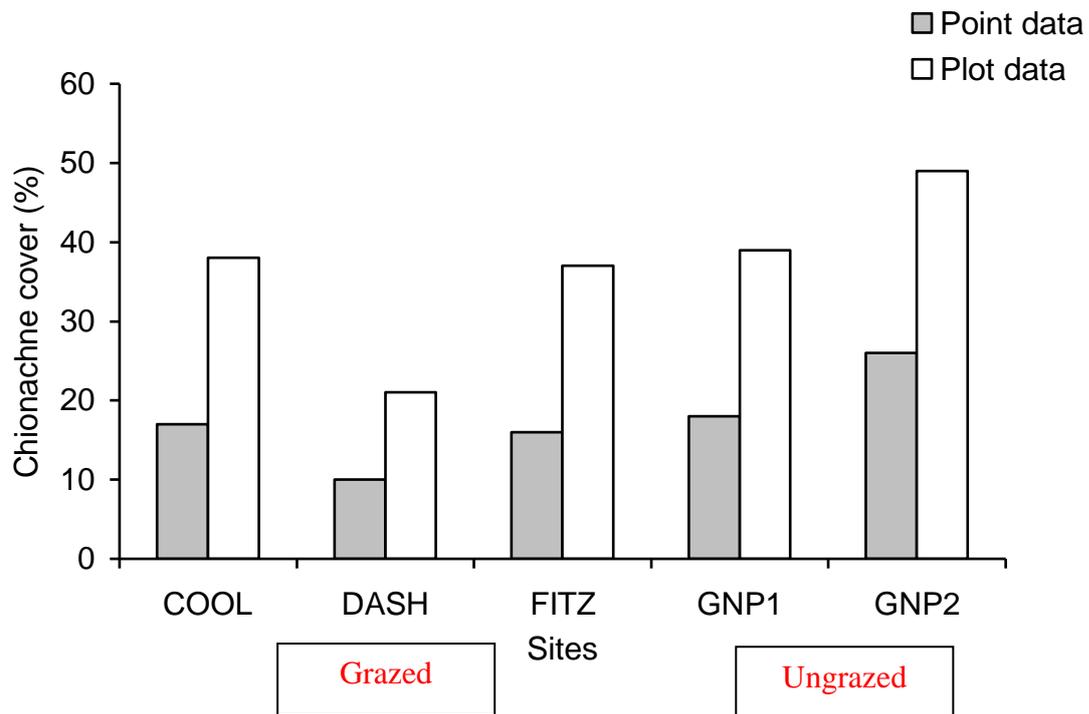


Figure 4-8. Frequency of *Chionachne* cover (%) at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ= Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

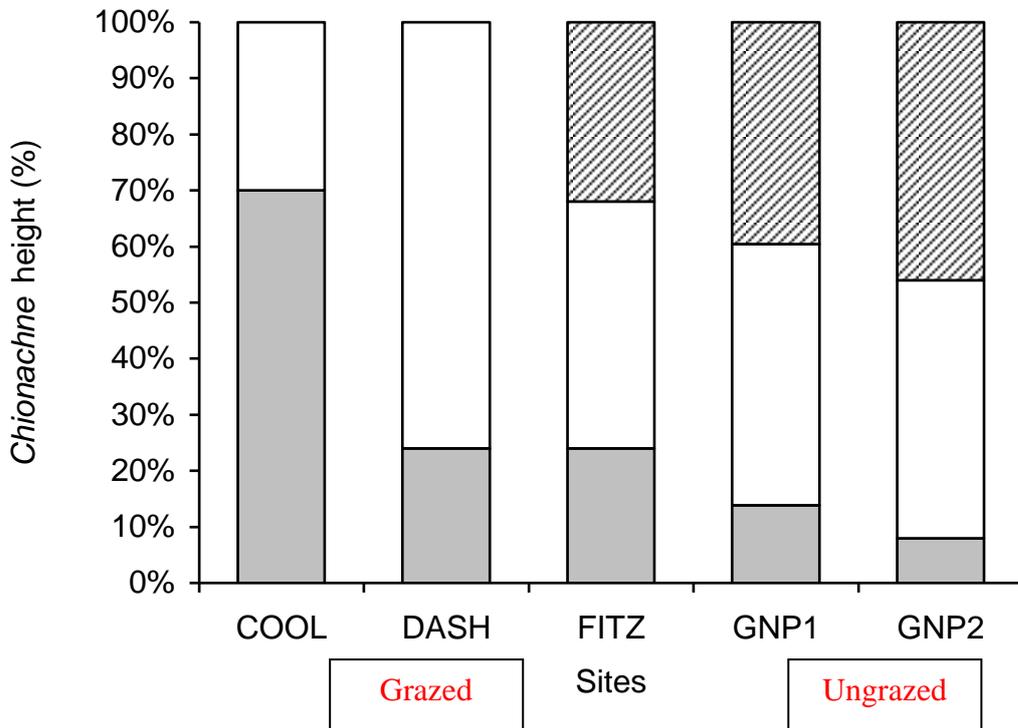


Figure 4-9. Height frequency distribution (%) of *Chionachne* at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

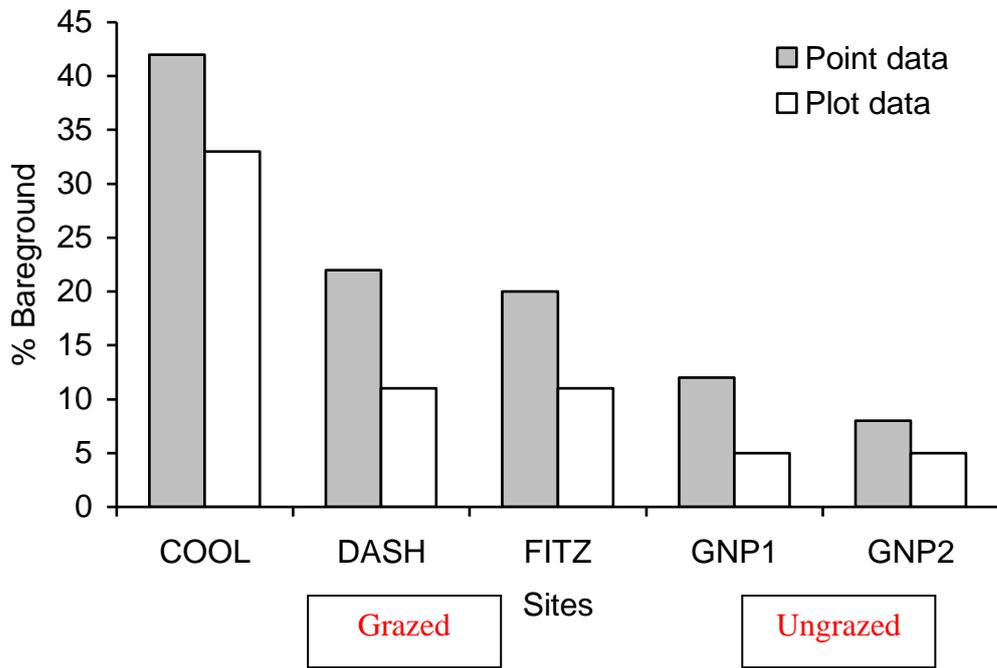


Figure 4-10. Frequency (%) of Bareground (BG) at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

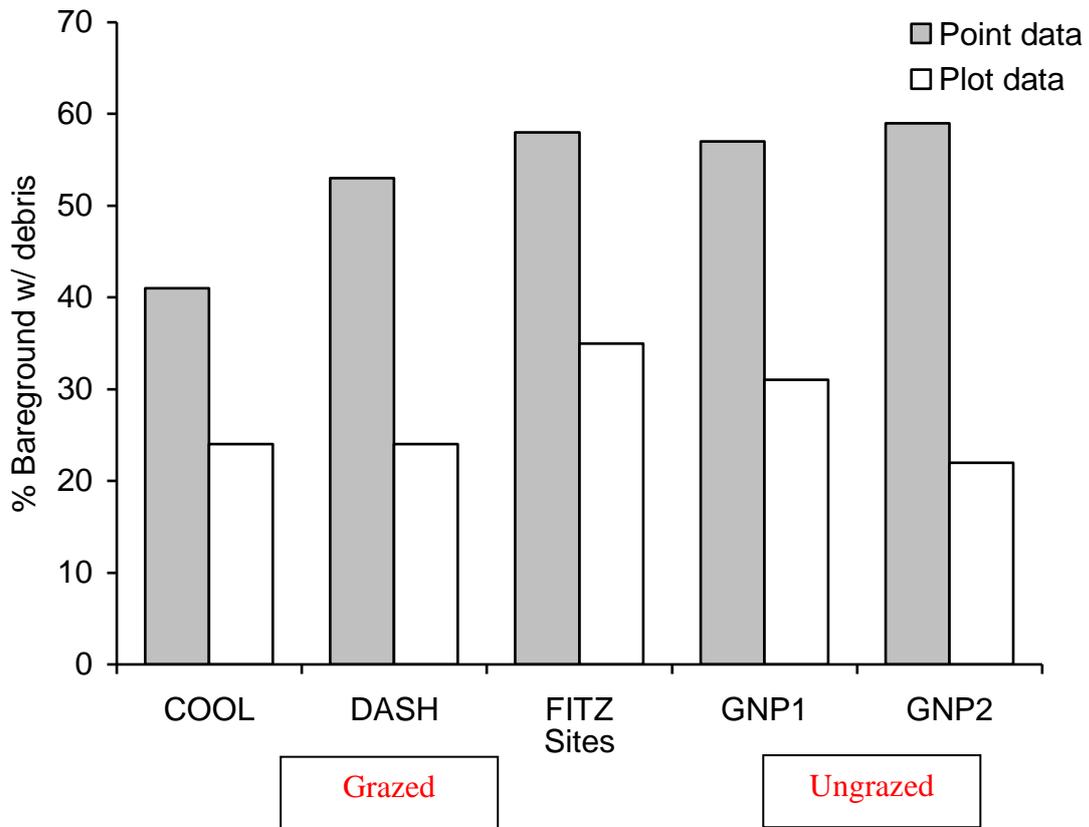


Figure 4-11. Frequency (%) of Bareground w/ debris (BGD) at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

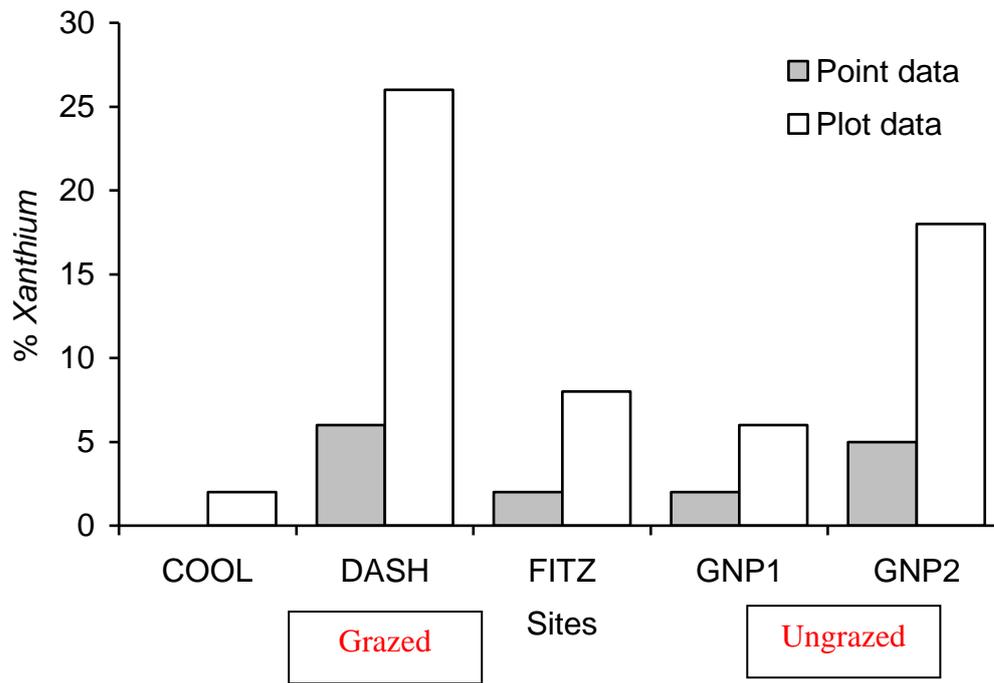


Figure 4-12a. Frequency (%) of *Xanthium* at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.



Figure 4-12b. River bank at Dashwood Crossing showing dense cover of *Xanthium*, August 2002 (*Xanthium* has already dried, though mid-way through the Dry Season).

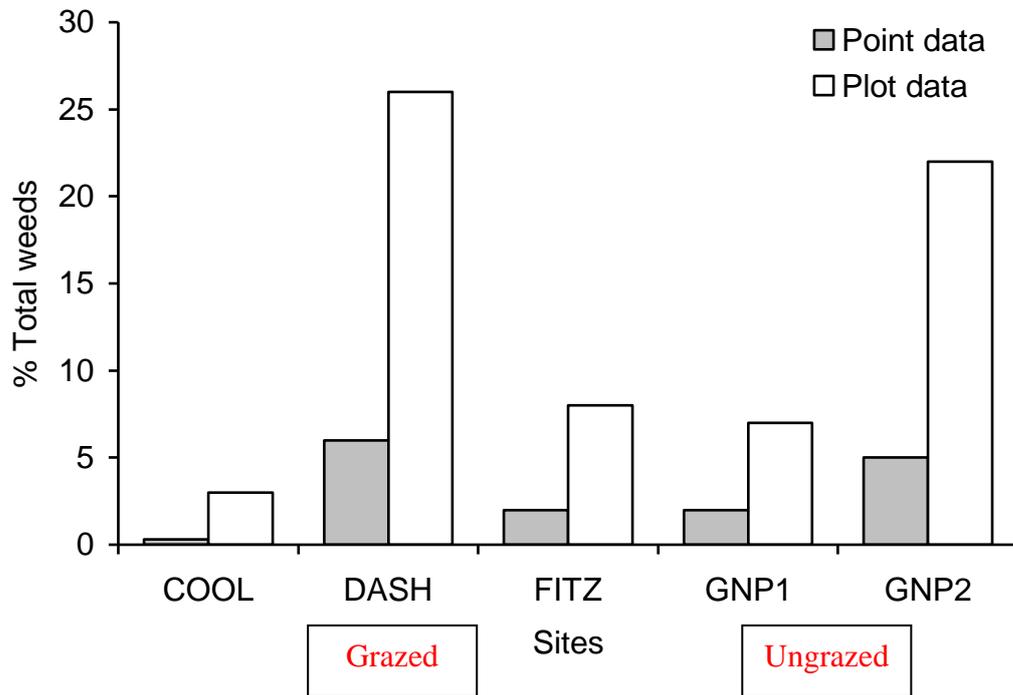


Figure 4-13. Frequency (%) of total weeds (*Xanthium*, *Ricinus*, and *Passiflora*) at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

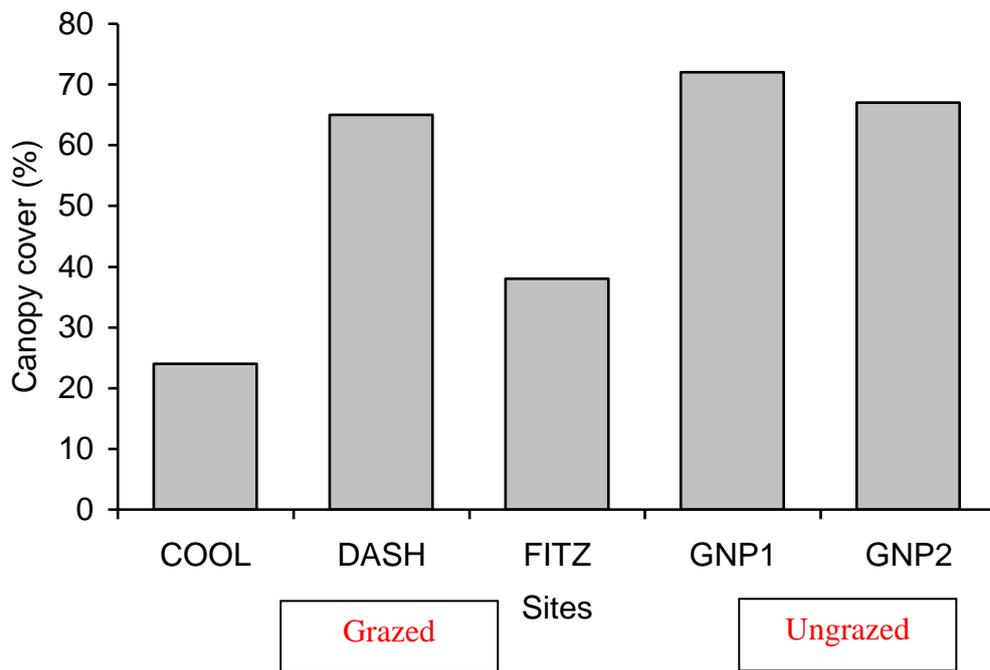


Figure 4-14. Canopy cover, expressed as a percentage at individual study sites (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

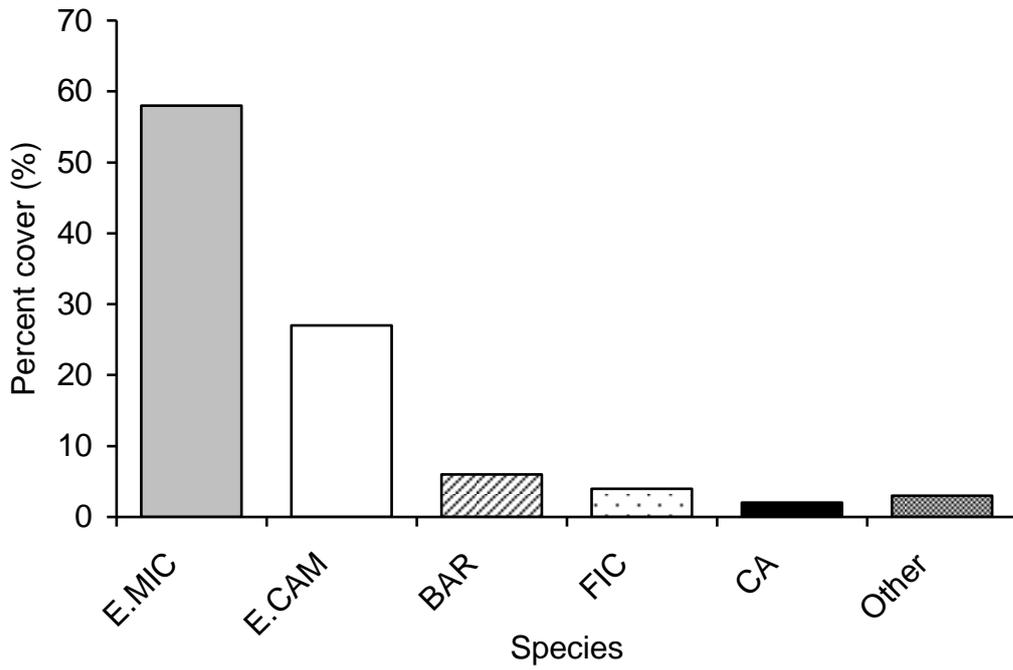


Figure 4-15. Canopy cover (species) at combined study sites, expressed as a percentage. E.MIC = *Eucalyptus microtheca*, E.CAM = *Eucalyptus camaldulensis*, BAR = *Barringtonia acutangula*, FIC = *Ficus coronulata*, CAT = *Cathormium umbellatum*. (n = 75).

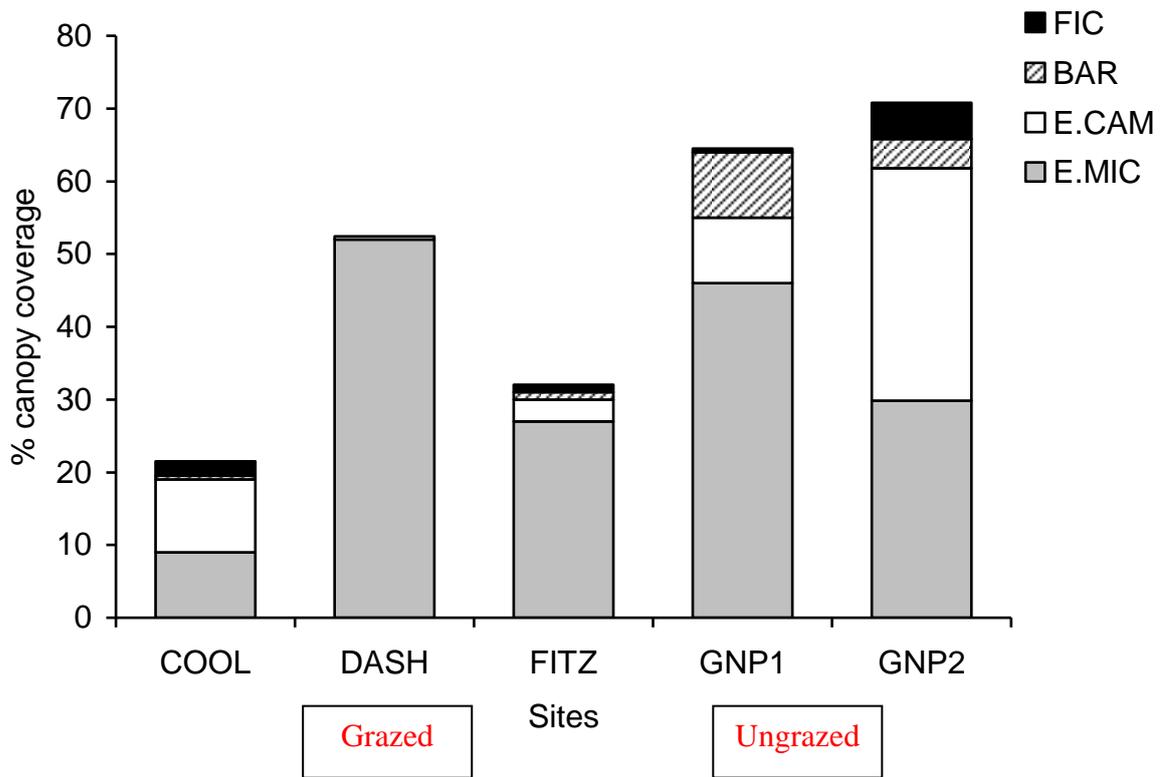


Figure 4-16. Dominant canopy cover composition (%) at individual study sites. E.MIC = *Eucalyptus microtheca*, E.CAM = *Eucalyptus camaldulensis*, BAR = *Barringtonia acutangula*, FIC = *Ficus coronulata* (n = 75). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

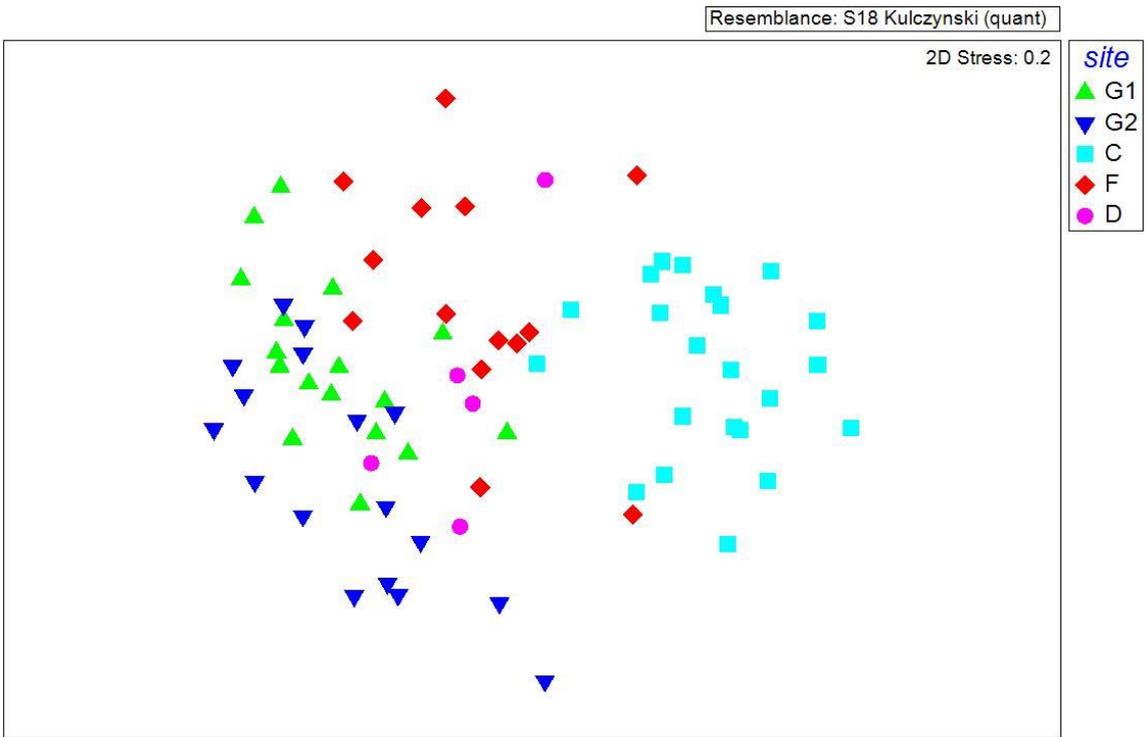


Figure 4-17. Ordination of transects, based on their overall similarity in habitat variables measured. Symbols mark the five separate study sites (C = Coolibah, D = Dashwood, F = Fitzroy, G1 = GNP1, G2 = GNP2).



Figure 4-18a. Fire at GNP1, 22 October, 2001. Picture taken approximately two hours post-fire.



Figure 4-18b. Three months post-fire at GNP1, January 31, 2002. Note that the broad-leaved vegetation is weeds (*Xanthium* and *Ricinus*) in addition to the dominant *Chionachne* seen clearly in the forefront.

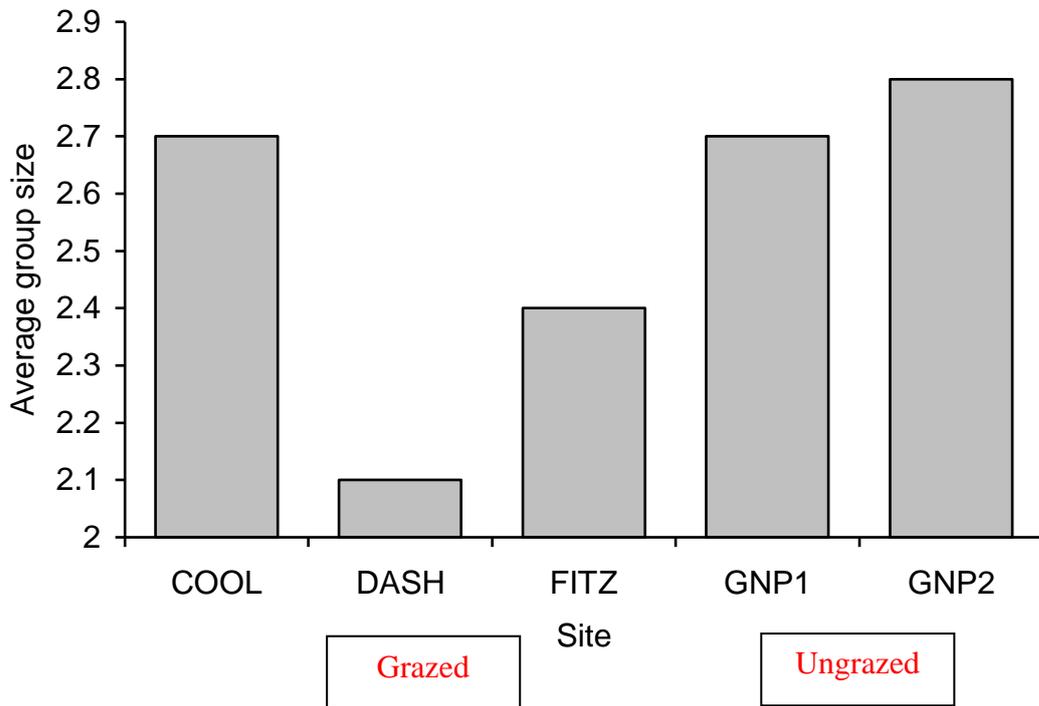


Figure 4-19. Average group sizes (adult birds only) at study sites 2001-2003 (n = 88 group-years). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites.

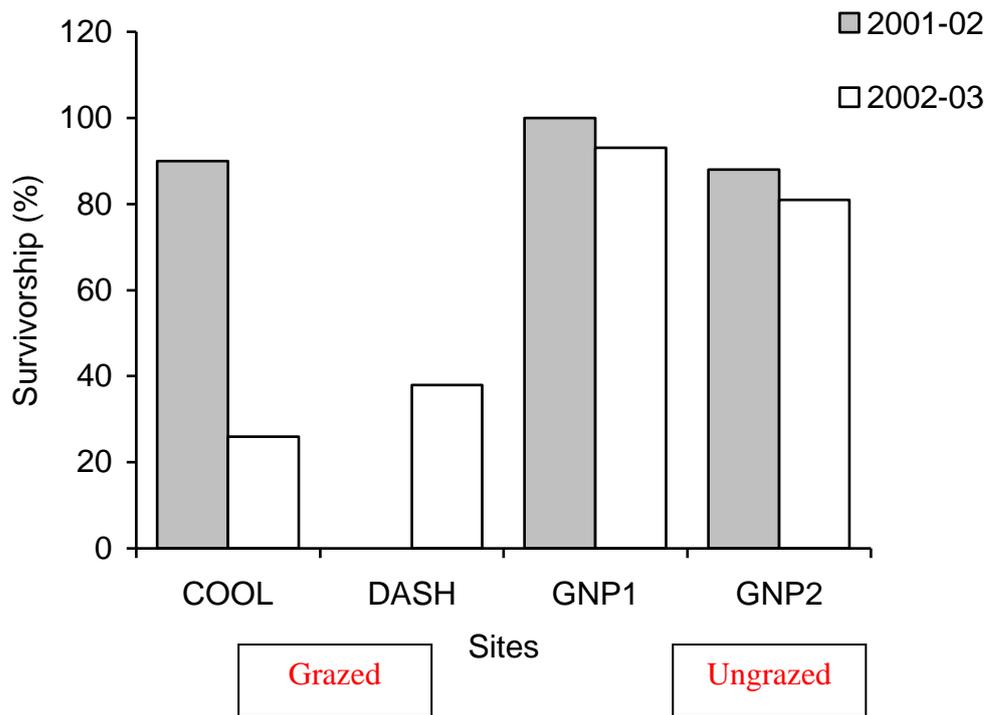


Figure 4-20. Adult survivorship at individual study sites from 2001-2002 and 2002-2003 (n = 40 group years). COOL = Coolibah, DASH = Dashwood, GNP1 = Gregory National Park Site 1, GNP2 = Gregory National Park Site 2. Note that survivorship at Dashwood was 0 from 2001 through 2002. Fitzroy was not included as visits were insufficient to accurately determine survivorship.

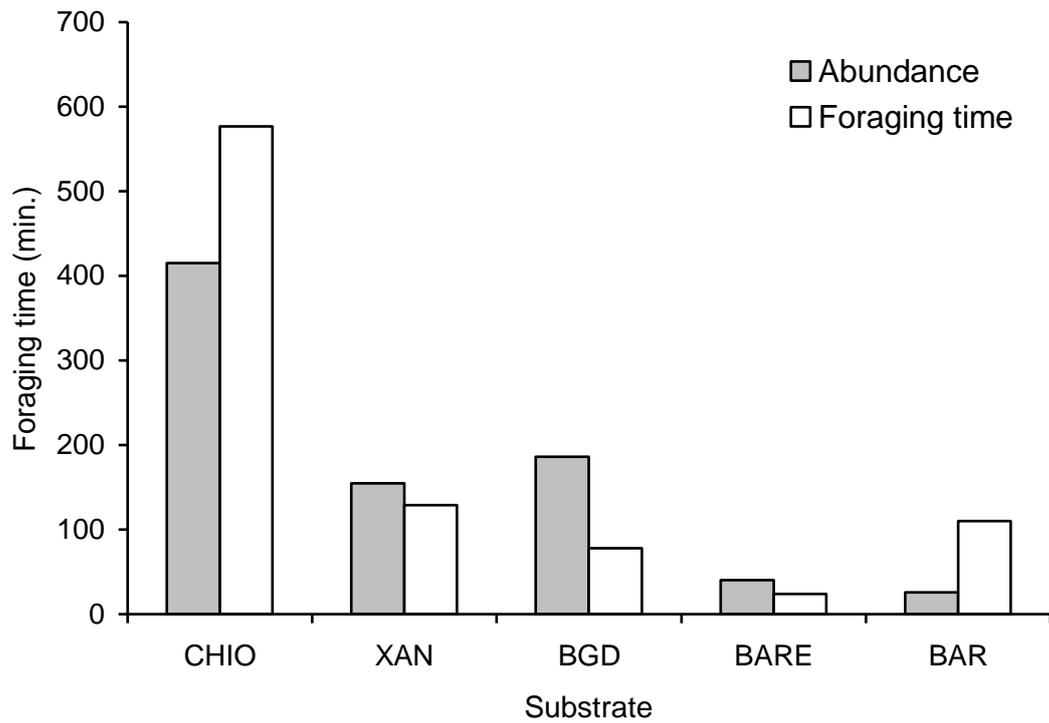


Figure 4-21. Foraging preference illustrated by time spent foraging in relation to substrate abundance (based on transect data-20 cm intercepts and minutes foraging). CHIO = *Chionachne*, XAN = *Xanthium*, BGD = Bareground w/ debris BARE = bareground, BAR = *Barringtonia*.

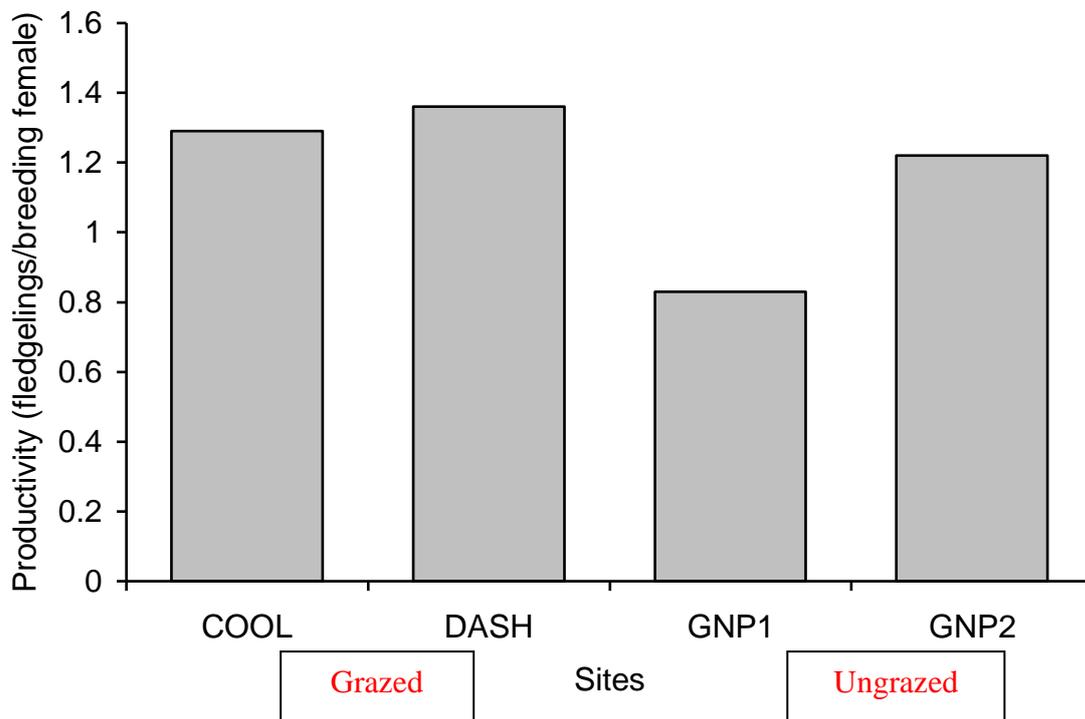


Figure 4-22. Productivity (number of fledglings per breeding female) at individual study sites, 2001-2003 (n = 80 group-years). COOL = Coolibah, DASH = Dashwood, FITZ = Fitzroy, GNP1 and GNP2 = Gregory National Park sites. Note that this graph represents productivity per bird but does not reflect the total number of fledglings produced or the total number of breeding females present at each site.

CHAPTER 5  
LANDSCAPE SCALE ANALYSES OF HABITAT AND THE IMPLICATIONS  
FOR CONSERVATION OF THE PURPLE-CROWNED FAIRY-WREN

This final chapter presents information on the distribution of the primary habitat of the PCFW in the larger landscape, and discusses implications for future conservation management. This chapter concludes with management recommendations for the PCFW population in the VRD in addition to future research directions.

**Introduction**

Indicator and/or focal species have been used in conservation biology in various contexts although their utility has been the subject of substantial debate (Ricketts *et al.*, 1999; Lindenmayer *et al.*, 2002; MacNally and Fleishman, 2004; Bani *et al.*, 2006). However, Lawler *et al.* (2003) found that at-risk species functioned well as an indicator group in by themselves. Their results concurred with Groombridge (1992), suggesting that managing and conserving at-risk species could help preserve biodiversity as a whole.

The focal species approach uses species most sensitive to a particular threat, for example, habitat specialists, and can be an important conservation tool (Lambeck, 1997). The utility of this approach was demonstrated by a study of woodland birds in southeastern Australia where conservation of the habitat of two focal species was shown to also provide for the needs of at least 95% of other woodland birds (Watson *et al.*, 2001). Because the PCFW is a riparian obligate, highly dependent on *Chionachne* in the VRD, it is reasonable to examine the wider distribution of this grass as an important indicator of the conservation potential of the PCFW itself as an indicator species for riparian zones. Lambeck (1997) included three categories of focal species: area- or habitat-limited species, movement-limited species, and

management-limited species. Based on its ecology, distribution, and response to grazing, the PCFW fits all of these categories making it useful indicator species. However, as the PCFW along the Victoria River is confined to a single understory grass species it may be more useful to include a host of riparian obligate species as a focal group in order to ascertain riparian health.

Habitat fragmentation, defined here as “the breaking apart of habitat, independent of habitat loss” (Fahrig, 2003), can affect community composition and structure (Wiens, 1989). Habitat fragmentation results in isolated patches of habitat in a variegated landscape (Andr n, 1994; Hobbs, 2005). The effects of fragmentation are often studied in relation to patch size and patch isolation, with models often based on a binary simplification of the environment into suitable habitat and “matrix” (unsuitable lands) (e.g. Hill and Caswell, 1999; Debinsky and Holt, 2000). However, fragmentation studies that have more biological realism (i.e. species-specific attributes and habitat quality) will most likely lead to much clearer conclusions (e.g. Debinsky and Holt, 2000; Cooper and Walters, 2002; Wiegand *et al.*, 2005). For example, in the woodlands of south-eastern Australia, the response of species to fragmentation varied in different landscapes, suggesting that the matrix plays can also play an important role (Watson *et al.*, 2005).

The effect of habitat fragmentation has been studied for several *Malurus* species (e.g. *Malurus pulcherrimis* and *Malurus splendens*) in Australia (Brooker and Brooker, 1994; Brooker and Brooker, 2001). For these malurids, Brooker and Brooker (2001) found that smaller patches had higher adult mortality rates than larger patches, but that reproductive success was higher in smaller patches (perhaps due to differences in brood parasitism rates). The authors concluded that both small and large patches were important for the conservation of *Malurus pulcherrimus*. In a

model developed to predict the response of *Malurus splendens* to fire and fragmentation, Brooker and Brooker (1994) found that the likelihood of populations going extinct in patches was highly dependent on fire. That is, if there is a high risk of wildfire, very large patches were needed to sustain the population but if fire were excluded, patches size less than a quarter the size of a large patch could be viable.

The strict adherence of the PCFW to *Chionachne* habitat along the Victoria River coupled with the potential impact of grazing on its primary habitat makes the PCFW especially prone to fragmentation effects. Long-distance dispersal capabilities of the PCFW are most likely limited, however, the wrens possibly have adaptations to natural disturbances such as flooding which, necessitate frequent local movements. The ability to disperse short distances from local disturbances may benefit the PCFW when it is necessary to respond to other novel habitat disturbances (e.g. grazing). The variable nature of the PCFW habitat suggests that the structure of non-*Chionachne* habitat may be crucial in any dispersal movements and that the conservation of non-*Chionachne* habitat may also play an important role in PCFW conservation (e.g. understory cover), as dispersal must occur over these non-*Chionachne* areas.

### **Canegrass (*Chionachne cyathopoda*) Distribution**

#### **Methods**

In order to determine the abundance of *Chionachne* in the broader landscape, and the degree of fragmentation of PCFW habitat, an estimate of *Chionachne* abundance and distribution along the Victoria River was required. Existing aerial photography was examined but as it was predominantly black and white, and it was impossible to distinguish *Chionachne* from adjacent vegetation. Additionally, most of the photography was also outdated. Satellite imagery was also examined but the resolution was too coarse a scale to accurately identify *Chionachne*. Higher

resolution scenes, which may have diagnostically shown *Chionachne* patches, were unaffordable. An opportunity to examine *Chionachne* via helicopter was made available, and this proved to be a very efficient and effective method of mapping *Chionachne*. I conducted mapping in August 2003, a time of year when *Chionachne* is clearly visible and distinct as it stays greener for a longer period than other ground vegetation components.

Due to funding constraints, only a limited area could be surveyed. I decided to focus on the main river course and cover as much area as possible that potentially could contain PCFW habitat. Earlier surveys by boat indicated that *Chionachne* does not extend much further downstream from the town of Timber Creek (15.631181 S, 130.47612 E), where the Victoria river is subject to tidal influence (and hence the nature of the riparian vegetation changes), and so this was chosen as the starting point. The end point of the survey, approximately 170 km upstream and the limit of the distance that could be covered, was Dashwood Crossing (16.33771 S, 131.10987 E), above which the Victoria River becomes a much smaller watercourse.

Both sides of the river were surveyed simultaneously using two observers (A. van Doorn and H. Henri), one on each side of the helicopter. I conducted a preliminary survey to determine feasibility and parameters (width and density of *Chionachne*) to be used for mapping *Chionachne* distribution.

Sections along the river where *Chionachne* occurred were measured for length along the river using hand-held GPS (Garmin GPS II Plus, <15m). The sections were not entirely continuous, of course, so patches of *Chionachne* were also mapped. For each patch, the starting and end points were marked and the following two variables were recorded; density (sparse, medium, dense) and width (narrow, medium, wide). Although it was impossible to estimate exact width using this method, I was able to

assign approximate width categories based on comparing known (measured) widths at field sites with estimates from the preliminary survey, as follows: narrow = <50m, Medium = 50-100m, and wide = 100-200m. Observers were calibrated during an initial trial. It was not possible to adequately survey tributaries that may have held additional *Chionachne*, although where visible *Chionachne* extended into the tributary were classified as being wide (i.e. 100-200m).

The area of each *Chionachne* patch was estimated by multiplying patch length with estimated patch width. For *Chionachne* area calculations, I used the upper limit for each width parameter, that is, narrow = 50 m, medium = 100m and wide = 200, and hence estimated total area represents the upper limit of the estimate of *Chionachne* area. Distributional maps of *Chionachne* were created using ESRI ARCGIS 9.1.

I estimated PCFW total population size for the Victoria River from Timber Creek through to Dashwood Crossing by dividing the total *Chionachne* area by the average area occupied by individual birds at study site (based on density data, Chapter 3). Assumptions underlying these estimates included: 1. *Chionachne* is the only primary habitat of the PCFW in this system (an assumption corroborated by substantial searches of this system in non-*Chionachne* habitat), 2. Small and fragmented patches were inhabited by PCFWs, as well as the larger patches (as were studied in detail on the ground), 3. PCFW density does not vary with *Chionachne* density (sparse, medium and dense).

## **Results**

Along the approximately 170 km stretch of river surveyed, the total area of *Chionachne* based on the widths assigned to each patch size was 1,222 ha. However,

the *Chionachne* distribution was severely fragmented into a total of 52 patches, with varying length, widths and densities (Figs. 5-1, 5-2).

The average patch size was 24.5 ha ranging from 0.23 ha to 244 ha (Table 5-1). The average distance from a patch to its nearest neighbor was 1.37 km, ranging from 0.33 to 41 km. Most patches, regardless of size, were near others, but there were also a substantial number of smaller isolated patches (Fig. 5-3).

*Chionachne* distribution was not equally distributed along the river and appeared to vary, especially between areas under different tenure (i.e. according to land use) (Figs. 5-1, 5-2, Table 5-1). Of the total amount of *Chionachne* found along the river, the largest percentage of *Chionachne* was found within Gregory National Park (40%) and within this area *Chionachne* was also the least fragmented (Figs. 5-1, 5-2). Further, in Gregory National Park, *Chionachne* had the largest average patch size by far, more than twice as large as that of the two pastoral properties (Table 5-1).

The largest number of small isolated patches was found on Bradshaw (this area had been pastoral property but was taken over by the defense department in 1996), nevertheless, this area still has a substantial amount (13%) of the total *Chionachne* area (Table 5-1). Fitzroy Station, also used for pastoralism, had a large area of *Chionachne* (32% of the total found) but was highly fragmented in areas (primarily downstream). Coolibah Station had several larger patches and also contained a considerable abundance of *Chionachne* (14% of the total area), particularly in the area adjoining the national park. The smallest amount of total *Chionachne* surveyed was found at Coolibah Crocodile Farm (1%) mainly because it consists of a very small area (Table 5-1).

One of the most interesting findings was the lack of suitable habitat for approximately 40 kilometers, from the last patch of *Chionachne* in Gregory National

Park, south to Dashwood Crossing on Victoria River Downs. This area is accessible only by air and hence prior to this survey, the extent of *Chionachne* in this stretch was unknown. A distance of 40 km is a significant barrier for a small understory bird with weak flight and strong habitat preferences, thus effectively isolating these two areas from one another (e.g. Brooker and Brooker, 1997).

Only one small patch of *Chionachne* was found on Victoria River Downs and this was located exactly at the end point of my survey (Dashwood Crossing). Additional areas of *Chionachne* do exist past the survey limit at Dashwood Crossing point but are apparently very patchy in distribution and do not occur in any great abundance (D. Hill, pers, comm.).

Average territory size of the PCFW was 0.41 ha (based on four sites, Coolibah, Fitzroy, GNP1 and GNP2; Chapter 3) and average group size was 2.6 (based 88 group-years; Chapter 3). Based on 1,222 ha total area of *Chionachne*, 2,955 family groups (or 7,713 individuals) could be present along this 170 km stretch of the Victoria River. Because the total area of *Chionachne* was used for this estimate of population size, it represents the “best case scenario”; however, it is extremely unlikely that the severely fragmented or very small patches would contain as high of densities of PCFWs as measured in Chapter 3, and the total population is actually much smaller. It is unlikely that a small patch isolated from other patches can sustain PCFWs in the long-term. In addition, as mentioned above, I used the upper limit of the width categories in calculating the total area of *Chionachne*, further ensuring a maximum size for the estimate of total PCFW population size. Also, the four study sites used here as a basis for average PCFW density estimates were all in moderate to high quality habitat and moderate to large patch size, so mean density across the broader habitat is probably appreciably less than this calculated figure.

To give an indication of how our estimate may be optimistic, if we consider that within a single season after cattle were introduced at Coolibah, the average group size dropped to 2.2 (Chapter 3), and if we applied this lower group size to the area of *Chionachne* used for pastoralism, the optimistic estimate of the total population size (above) would decrease by 712 birds (9%). Further, if we apply the fact that the number of birds at Coolibah decreased by 64% in one season when cattle were introduced to the site (Chapter 3), and apply this percentage to the total area of *Chionachne* used for pastoralism, then the total population estimate for the PCFW would be 2,283 (30%) fewer birds than the “best scenario” estimate, a new total of 5,430 birds.

Despite the uncertainty of the population size estimate based on the *Chionachne* survey, the distribution is very useful for comparative measures to identify key areas for conservation and differences between land uses. The distribution map of *Chionachne* from my study

### **Conservation and Management Recommendations**

Purple-crowned Fairy-wrens depend on *Chionachne* for foraging, breeding and cover and hence any conservation action should focus on *Chionachne* preservation.

Below I have outlined management recommendations based on findings from my study, with each potentially threatening process being addressed individually. Grazing is dealt with explicitly as my study focused on this particular element. Fragmentation of habitat is also dealt with as a threatening process, understanding that fragmentation is most likely the product of several factors. Based on my own experiences with this project, I have outlined some issues and strategies to address stakeholder involvement, a key issue in a multiple land use area such as the VRD.

Finally, I have also included a section on future research directions as a guidance that may serve as a guide to other researchers or government agencies.

### **Grazing**

Based on results from my study, grazing is one of the most important threats to the PCFW in the VRD. Vegetation at grazed sites offered far less cover for PCFWs than ungrazed sites, primarily due to the difference in *Chionachne* height (Chapter 4). Of greatest significance was the alarming increase in adult mortality, resulting in fewer breeding pairs, within one season at the newly intensely-grazed site, Coolibah. Such a large increase in adult mortality over such a short period of time demonstrates the acute impact that grazing can have if it is conducted at high intensities. Unfortunately, survival rates at Fitzroy, which was grazed lightly, are not well known but *Chionachne* height was not compromised as much as at Coolibah, and it is reasonable to expect that survival rates might be more favorable at lower grazing pressures. The low incidence of *Chionachne* at the Dashwood site, in areas where *Chionachne* might be expected to occur (Chapter 4), suggests that long-term grazing may eventually lead to a loss of *Chionachne* stands themselves.

Fencing the entire riparian corridor along the Victoria River and tributaries would most likely provide the best conservation measure for the PCFW, however, it is extremely unlikely this will be accomplished, for a number of reasons. The remote location of the region and frequent flooding make fencing and maintaining fences over large areas very difficult, costly, and time consuming. For most stations, the cost of fencing make it economically unattractive, and perhaps in some cases, prohibitive. Nevertheless, fencing of the riparian corridor should be encouraged and with government subsidies these costs are much reduced. In all likelihood a combination of land management techniques will need to be incorporated to reach a compromise

and undoubtedly these will need to be integrated into a larger catchment conservation management plan

Grazing by feral animals (i.e. buffalo, pigs, scrub bulls) should also not be overlooked as these animals can also severely affect riparian vegetation. Control of feral animals will not only benefit riparian conservation but will also benefit pastoralism by reducing competition with cattle.

### **Weeds**

Weeds are widespread throughout the VRD. Weeds pose a threat not only to biodiversity (e.g. by competing with desirable native species), but also impact pastoralism by potentially sickening stock, reducing desirable pasture species and hampering access to water. In relation to the PCFW, weeds can be for cover and foraging, and are preferable to bareground. However, weeds are not a preferred substrate, at least in the case of *Xanthium* where *Chionachne* is available, nor do birds nest in weeds (Chapter 4). The control of most weeds in the district is not plausible and attention should focus on the eradication of new invasive species before they can spread.

### **Fire**

Despite the limited data available for the effects of fire on *Chionachne* and PCFW, we can infer from the width and connectivity of *Chionachne* in some areas (e.g. Gregory National Park), that a late season (hot) burn could cover a very large area and have severe consequences for the PCFW (Chapter 4). Although *Chionachne* regenerates rapidly after fire, a wide scale fire could result in high mortality rates of the PCFW. Firebreaks would provide some protective measures in areas of high connectivity. However, management actions will need to consider the potential of

larger fire breaks to further fragment habitat and that fire breaks may become infested with weeds.

### **Fragmentation**

The survey conducted during the course of my study has demonstrated the fragmented nature of *Chionachne* distribution. Because the largest aerial extent of *Chionachne* is within Gregory National Park, from a conservation standpoint this area is critical for the long-term survival of the PCFW. Both Coolibah Station and Fitzroy Station directly abut the national park, and have considerable amount of *Chionachne*, including larger patches that are separated by only small distances. Small patches of *Chionachne* located between larger patches would ideally be conserved to increase connectivity.

Observations made during my study indicated that there is a dispersal threshold beyond which re-colonization is unlikely. Two large patches of dense but isolated, *Chionachne* were found in the Victoria River catchment that had no PCFWs present. Conversations with land managers revealed that these patches of *Chionachne* had only recently regenerated after having been grazed intensely for a long time. Potentially, these areas of suitable habitat could be used as re-introduction sites once the habitat suitability is determined. Future research to determine dispersal threshold, in combination with an investigation into dispersal habitat suitability, will allow for more accurate management actions to determine patches of high conservation value. I suggest that the dispersal threshold will not be very high, judging from PCFW behavior and morphology.

### **Stakeholder Involvement**

In remote location such as the VRD, garnering community support may be a pivotal element for any long-term success. Purple-crowned fairy-wrens are

distributed in multiple land use areas, making stakeholder involvement increasingly important as connectivity undoubtedly plays a major role in long-term PCFW conservation. In the case of the PCFW, there are multiple avenues for stakeholder involvement to bolster support for any required conservation action. Government subsidies for conservation management priorities (fencing, weed and erosion control) are extremely important for the future management of riparian zones in the VRD and full use should be made of these funds. The Victoria River Conservation Association (VRDCA) can make an important contribution to any future conservation efforts by continued efforts in funding applications and liaison with stakeholders. Notably, the PCFW is a good candidate for such communication and conservation action, as it is an aesthetic and high profile species, well known to landholders and other stakeholders.

Ecological knowledge alone will not be sufficient to develop or implement an adequate management plan. Without participation of, and cooperation among, the stakeholders, the habitat will remain fragmented and the region will continue to have small isolated populations of PCFW. In addition, because barriers are often lacking between adjacent properties there are always flow-on effects from one property to the next. For example, if one stakeholder is actively controlling weeds, but the property upstream is not, then the effectiveness of weed control will always be compromised. Although this point may seem logical and straightforward, it emphasizes the need for a long-term commitment and continuous stakeholder involvement.

### **Conservation Management Recommendations**

Below I have outlined a series of management recommendations that should be included in the development for a conservation management plan for the VRD.

#### **A. Population Estimation and Monitoring Activities**

- Identify key areas for conservation (i.e. areas that will increase connectivity by linking patches) and instigate conservation methods at these sites (e.g. fencing, erosion control).
- Sampling *Chionachne* patches of varying size and isolation, to determine PCFW absence or presence in addition to determining PCFW densities in accordance with habitat quality, allowing for a more accurate population estimate.
- A widespread survey of the Victoria River catchment to determine PCFW distribution and habitat associations.
- Establishment and maintenance of population monitoring, where possible linked to assessment of efficacy of management inputs.

#### B. Land Use and Habitat Assessment Activities

- Preservation of healthy *Chionachne* stands (fencing, fire breaks).
- Control of feral animals across all sites.
- Amelioration of grazing pressure in riparian areas using a variety of methods including reduced stocking rates, increased artificial water sources, fencing core areas of *Chionachne*, and restricted access.
- Erosion control measures to avoid losing *Chionachne* root bases.
- Long-term monitoring of vegetation monitoring to assess habitat condition.

#### C. Stakeholder, Public, and Government Involvement

- Enhanced development of stakeholder involvement (regular contact, dissemination of information, involvement in decision making).
- An assessment of fencing costs in cooperation with stakeholders.
- Enhanced communication and publicity to increase public awareness and interest to engender community support.
- At a broader scale, liaison with government and research agencies in Western Australia for collaborative research and management strategies.

#### **Future research directions**

- Investigation into nest predation rates and causes.
- Research on dispersal dynamics (e.g. maximal dispersal distances, dispersal mortality, dispersal habitat use).
- Further investigation into PCFW ecology (e.g. paternity, incest avoidance).

- Metapopulation theory applications to examine population dynamics of the PCFW.
- Population viability analyses (PVA) to investigate disturbance effects and develop predictive models.
- Development of behavior-based (based on fitness maximization) population models which may produce more accurate predictions.
- Investigation into the role of *Chionachne* as an erosion control agent.
- Further investigations into the role of weeds, fire, grazing and other disturbances (e.g. interactions, confounding effects).
- Further research into pastoral management in riparian zones (e.g. stocking rates, resting periods, exclusion methods) that is more compatible with preserving *Chionachne*.

### **Conclusion**

Although many aspects of the ecology of the PCFW and identification of threatening processes were examined during the course of my study, there are still many areas that need further investigation. As is the case with most studies, answering one question leads to many more. However, the results found during my study have provided a substantial basis for conservation management initiatives. Common sense conservation initiatives, such as riparian corridor protection, should be initiated as soon as possible, using coordinated efforts combining solid science, community and stakeholder education and participation, and if necessary using any legislation available by government agencies to protect this unique species and its distinctive habitat.

It is possible to continue sitting on the fence, touting the need for more data, however, that stance will do nothing to conserve a species with a fragmented distribution whose habitat needs immediate protection. We need to implement conservation strategies to conserve existing riparian habitat, and in so doing, will preserve not only the PCFW, but potentially also other dependent riparian species.

The other alternative, rehabilitation at some later date, is an expensive proposition in an area as vast as the Victoria River District.

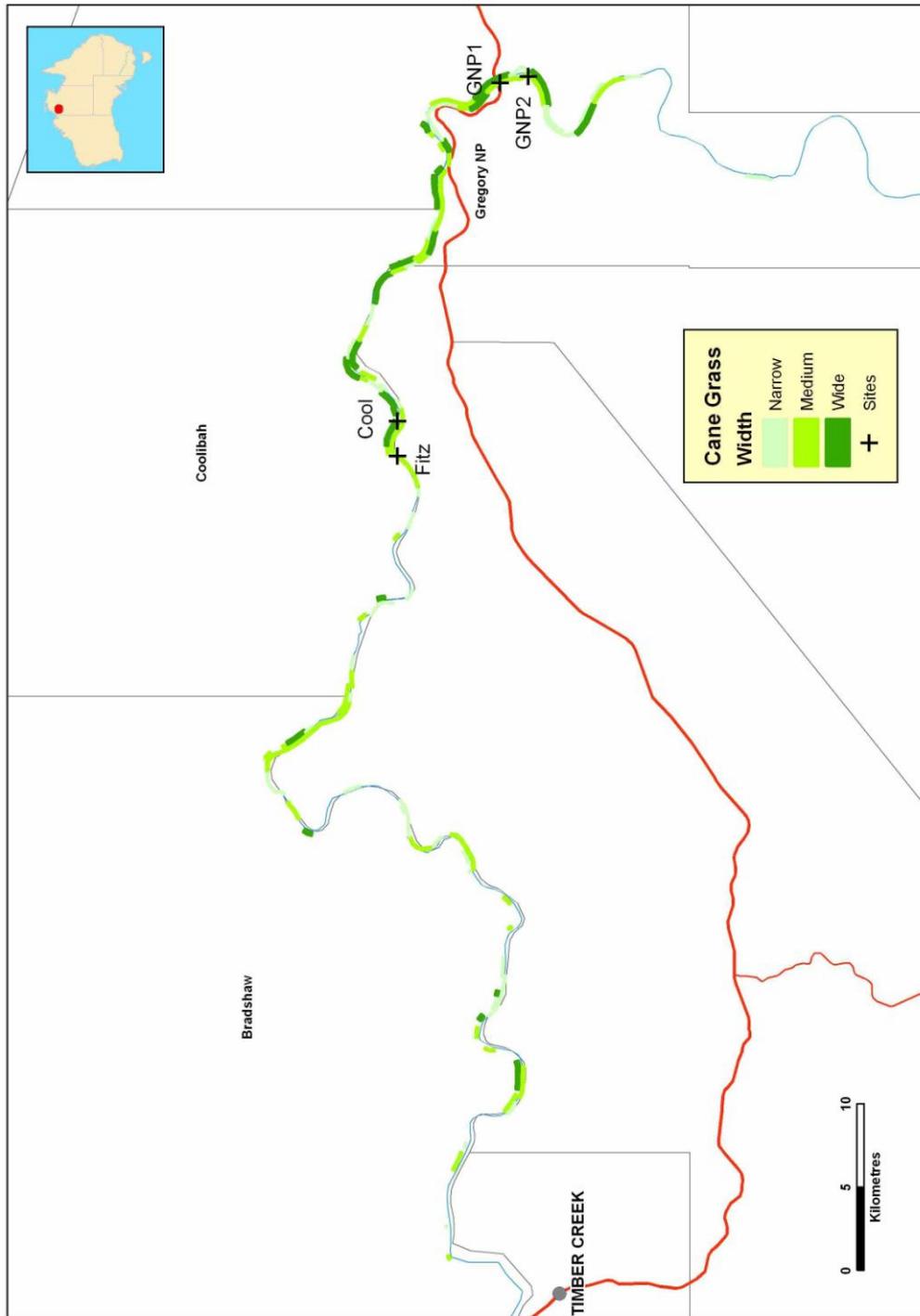


Figure 5-1. Map showing width of canegrass (*Chionachne cyathopoda*) along the Victoria River. Note that this map does not include one study site, Dashwood Crossing south of Gregory National Park, as the next patch of *Chionachne* was located at the end of the survey.

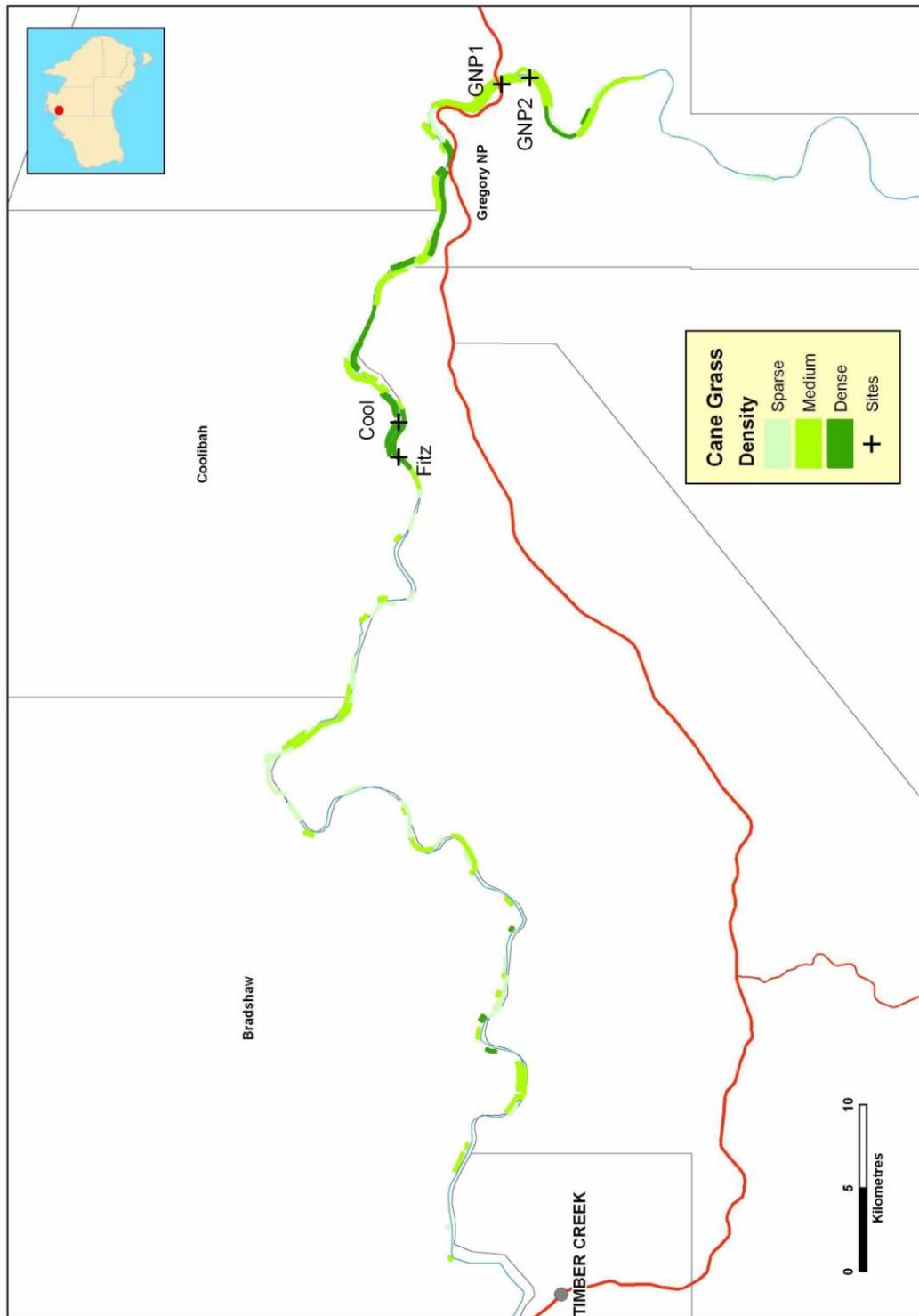


Figure 5-2. Map showing the density of canegrass (*Chionachne cyathopoda*) along the Victoria River. Note that this map does not include one study site, Dashwood Crossing south of Gregory National Park, as the next patch of *Chionachne* was located at the end of the survey.

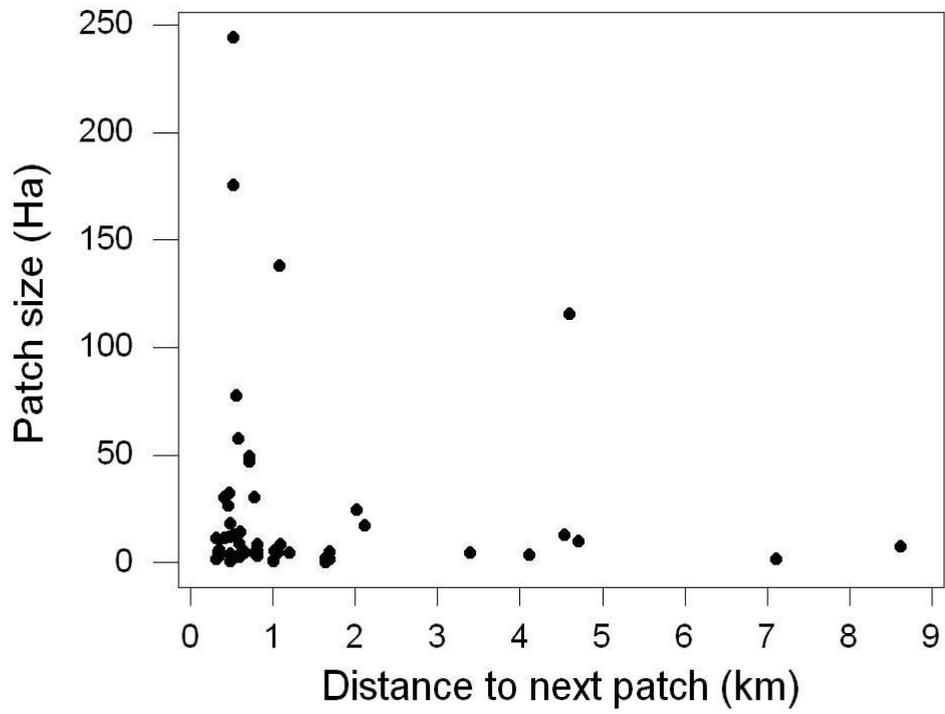


Figure 5-3. Canegrass (*Chionachne cyathopoda*) patch size and distance to next patch (n = 52).

Table 5-1. Canegrass (*Chionachne cyathopoda*) distribution, patch size and patch isolation by tenure (from helicopter based survey from Timber Creek to Dashwood Crossing, not including tributaries, based on canegrass width approximations). Note: Bradshaw was a pastoral property prior to 1996.

Site name	Tenure	Land use	Total Canegrass area (ha) (%)	No. of patches	Ave. patch size (ha)	Ave. distance to next patch (km)	PCFW conservation value	Potential implementation of conservation
Gregory National Park	Conservation	Conservation	486.09 (40%)	7	69.44	1.74	High	High
Fitzroy Station	Aboriginal	Pastoral (varied across area)	389.36 (32%)	16	24.33	0.99	High	Low/medium
Coolibah Station	Pastoral lease	Pastoral	175.65 (14%)	6	29.27	1.93	High	Low
Bradshaw	Defense	Defense training	153.56 (13%)	21	7.31	1.22	Medium	High
Coolibah Crocodile Farm	Private	Private (crocodile farming)	17.32 (1%)	2	17.32	1.93	Medium	High

APPENDIX A  
BIRD SPECIES

Scientific name	Common name
<i>Coturnix ypsilophora</i> (Bosc)	Brown quail*
<i>Dendrocygna eytoni</i> (Eyton)	Plumed whistling-duck
<i>Tadorna radjah</i> (Lesson)	Radjah shelduck
<i>Anas superciliosa</i> (Gmelin)	Pacific black-duck
<i>Anhinga melanogaster</i> (Pennant)	Darter
<i>Phalacrocorax melanoleucos</i> (Vieillot)	Little pied cormorant
<i>Pelecanus conspicillatus</i> (Temminck)	Australian pelican
<i>Egretta novaehollandiae</i> (Latham)	White-faced heron
<i>Ardea pacifica</i> (Latham)	White-necked heron
<i>Egretta garzetta</i> (Linnaeus)	Little egret
<i>Ardea Alba</i> (Linnaeus)	Great egret
<i>Ardea intermedia</i> (Wagler)	Intermediate egret
<i>Nycticorax caledonicus</i> (Gmelin)	Nankeen night heron
<i>Platalea regia</i> (Gould)	Royal spoonbill
<i>Platalea flavipes</i> (Gould)	Yellow-billed spoonbill
<i>Ephippiorhynchus asiaticus</i> (Latham)	Black-necked stork
<i>Aviceda subcristata</i> (Gould)	Pacific baza
<i>Elanus axillaris</i> (Latham)	Black-shouldered kite
<i>Milvus migrans</i> (Boddaert)	Black kite
<i>Haliastur sphenurus</i> (Vieillot)	Whistling kite
<i>Haliastur indus</i> (Boddaert)	Brahminy kite
<i>Haliaeetus leucogaster</i> (Gmelin)	White-bellied sea-eagle
<i>Accipiter fasciatus</i> (Vigors & Horsfield)	Brown goshawk
<i>Accipiter novaehollandiae</i> (Gmelin)	Collared sparrowhawk
<i>Falco berigora</i> (Vigors & Horsfield)	Brown falcon
<i>Falco peregrinus</i> (Tunstall)	Peregrine falcon
<i>Falco cenchroides</i> (Vigors & Horsfield)	Nankeen kestrel
<i>Grus rubicunda</i> (Perry)	Brolga
<i>Actitis hypoleucos</i> (Linnaeus)	Common sandpiper
<i>Burhinus grallarius</i> (Latham)	Bush stone-curlew
<i>Elseynornis melanops</i> (Vieillot)	Black-fronted dotterel
<i>Vanellus miles</i> (Boddaert)	Masked lapwing
<i>Geopelia striata</i> (Linnaeus)	Peaceful dove*
<i>Geopelia humeralis</i> (Temminck)	Bar-shouldered dove*
<i>Calyptorhynchus banksii</i> (Latham)	Red-tailed Black-cockatoo
<i>Cacatua roseicapilla</i> (Vieillot)	Galah
<i>Cacatua sanguinea</i> (Gould)	Little corella*
<i>Cacatua galerita</i> (Latham)	Sulphur-crested cockatoo
<i>Trichoglossus haematodus</i> (Linnaeus)	Rainbow lorikeet
<i>Psitteuteles versicolor</i> (Lear)	Varied lorikeet
<i>Aprosmictus erythropterus</i> (Gmelin)	Red-winged parrot
<i>Chrysococcyx osculans</i> (Gould)	Black-eared cuckoo
<i>Chrysococcyx basalis</i> (Horsfield)	Horsfield's bronze-cuckoo
<i>Eudynamys scolopacea</i> (Linnaeus)	Common koel
<i>Scythrops novaehollandiae</i> (Latham)	Channel-billed cuckoo

<i>Centropus phasianinus</i> (Latham)	Pheasant coucal*
<i>Ninox connivens</i> (Latham)	Barking owl
<i>Ninox novaeseelandiae</i> (Gmelin)	Southern boobook
<i>Tyto alba</i> (Scopoli)	Barn owl
<i>Podargus strigoides</i> (Latham)	Tawny frogmouth
<i>Eurostopodus argus</i> (Hartert)	Spotted nightjar
<i>Aegotheles cristatus</i> (Shaw)	Australian owl-nightjar
<i>Alcedo azurea</i> (Latham)	Azure kingfisher
<i>Dacelo leachii</i> (Vigors & Horsfield)	Blue-winged kookaburra
<i>Todiramphus sanctus</i> (Vigors & Horsfield)	Sacred kingfisher
<i>Merops ornatus</i> (Latham)	Rainbow bee-eater
<i>Eurystomus orientalis</i> (Linnaeus)	Dollarbird
<i>Climacteris melanura</i> (Gould)	Black-tailed treecreeper
<i>Malurus coronatus</i> (Gould)	Purple-crowned fairy-wren*
<i>Malurus melanocephalus</i> (Latham)	Red-backed fairy-wren*
<i>Pardalotus striatus</i> (Gmelin)	Striated pardalote*
<i>Smicrornis brevirostris</i> (Gould)	Weebill
<i>Philemon buceroides</i> (Swainson)	Helmeted friarbird
<i>Philemon argenticeps</i> (Gould)	Silver-crowned friarbird
<i>Philemon citreogularis</i> (Gould)	Little friarbird
<i>Lichenostomus unicolor</i> (Gould)	White-gaped honeyeater*
<i>Lichenostomus flavescens</i> (Gould)	Yellow-tinted honeyeater*
<i>Melithreptus albogularis</i> (Gould)	White-throated honeyeater
<i>Lichmera indistincta</i> (Vigors & Horsfield)	Brown honeyeater*
<i>Ramsayornis fasciatus</i> (Gould)	Bar-breasted honeyeater
<i>Conopophila rufogularis</i> (Gould)	Rufous-throated honeyeater
<i>Certhionyx pectoralis</i> (Gould)	Banded honeyeater
<i>Microeca fascinans</i> (Latham)	Jacky winter
<i>Pomastostomus temporalis</i> (Vigors & Horsfield)	Grey-crowned babbler
<i>Daphoenositta chrysoptera</i> (Latham)	Varied sitella
<i>Colluricincla harmonica</i> (Latham)	Grey shrike-thrush
<i>Myiagra rubecula</i> (Latham)	Leaden flycatcher
<i>Myiagra alecto</i> (Temminck)	Shinning flycatcher
<i>Myiagra inquieta</i> (Latham)	Restless flycatcher*
<i>Grallina cyanoleuca</i> (Latham)	Magpie-lark
<i>Rhipidura fuliginosa</i> (Sparman)	Grey fantail
<i>Rhipidura rufiventris</i> (Vieillot)	Northern fantail*
<i>Rhipidura leucophrys</i> (Latham)	Willie wagtail*
<i>Coracina novaehollandiae</i> (Gmelin)	Black-faced cuckoo-shrike
<i>Coracina papuensis</i> (Gmelin)	White-bellied cuckoo-shrike
<i>Lalage sueurii</i> (Vieillot)	White-winged triller
<i>Oriolus flavocinctus</i> (Vigors)	Olive-backed oriole
<i>Artamus leucorhynchus</i> (Linnaeus)	White-breasted woodswallow
<i>Artamus cinereus</i> (Vieillot)	Black-faced woodswallow
<i>Artamus minor</i> (Vieillot)	Little woodswallow
<i>Cracticus nigrogularis</i> (Gould)	Pied butcherbird
<i>Corvus orru</i> (Bonaparte)	Torresian crow*
<i>Chlamydera nuchalis</i> (Jardine & Selby)	Great bowerbird
<i>Taeniopygia bichenovii</i> (Vigors & Horsfield)	Double-barred finch*
<i>Poephila acuticauda</i> (Gould)	Long-tailed finch*
<i>Poephila personata</i> (Gould)	Masked finch*

*Neochmia phaeton* (Hombron & Jacquinot)  
*Neochmia ruficauda* (Gould)  
*Lonchura flaviprymna* (Gould)  
*Lonchura castaneothorax* (Gould)  
*Heteromunia pectoralis* (Gould)  
*Dicaeum hirundinaceum* (Shaw)  
*Cisticola exilis* (Vigors & Horsfield)

Crimson finch\*  
Star finch\*  
Yellow-rumped mannikin\*  
Chestnut-breasted mannikin\*  
Pictorella mannikin\*  
Mistletoebird  
Golden-headed cisticola\*

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Bird species observed at riparian study sites. Species marked with \* were found within *Chionachne cyathopoda* stands, all other species were located in the riparian habitat within 50m of *Chionachne cyathopoda* or in the canopy above it. Classification follows Christidis and Boles (1994).

APPENDIX B  
PLANT SPECIES

Botanical name	Common name
<i>Acacia holosericea</i> (A. Cunn.)	Candelabra wattle
<i>Barringtonia acutangula</i> (L.) Gaertn.	Freshwater mangrove
<i>Bauhinia cunninghamii</i> (Benth.) Benth.	Bauhinia
<i>Cathormiom umbellatum</i> (Vahl) Kostermans	Cathormiom
<i>Chionachne cyathopoda</i> (F. Muell.)	River grass (canegrass)
<i>Crotalaria novae-hollandiae</i> (DC)	New Holland rattlepod
<i>Eucalyptus camaldulensis</i> (Dehn.)	River red gum
<i>Eucalyptus microtheca</i> (F. Muell.)	Flooded box (coolibah)
<i>Excoecaria parviflora</i> (Muell. Arg.)	Gutta-percha tree
<i>Ficus coronulata</i> (F. Muell.)	River fig
<i>Flueggia virosa</i> (Roxb.ex Willd) Voigt	Dogwood
<i>Hyptis suaveolens</i> (L.)	Hyptis*
<i>Melaluca leucadendra</i> (L.)	Cadjeput
<i>Mnesithea rottboelioides</i> (R. Br.)	Northern canegrass
<i>Nauclea orientalis</i> (L.)	Leichardt pine
<i>Pandanus aquaticus</i> (F. Muell.)	River pandanus
<i>Parkinsonia aculeata</i> (L.)	Parkinsonia*
<i>Passiflora foetida</i> (L.)	Wild passionfruit*
<i>Ricinus communis</i> (L.)	Castor oil plant*
<i>Xanthium strumarium</i> (L.)	Noogoora burr*

List of plant species found within Purple-crowned fairy-wren habitat. Note that the list is based on vegetation transects within Purple-crowned fairy-wren territories (described in Chapter 3). Vegetation transects were conducted during the dry season resulting in absence some species (e.g. annual wet season grasses). Introduced species are indicated with an \*. Classification of botanical names follows Petheram and Kok (2003).

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Annamaria continues to live in the Northern Territory where she will be conducting further research on purple-crowned fairy-wrens and working as an independent environmental consultant.