

**Avifaunal ecology and responses to post-fire succession of
buttongrass moorlands in the
Tasmanian Wilderness World Heritage Area**



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Declaration of originality

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Statement of ethical conduct

The research associated with this thesis abides by the international and Australian codes on human and animal experimentation, the guidelines by the Australian Government's Office of the Gene Technology Regulator and the rulings of the Safety, Ethics and Institutional Biosafety Committees of the University.

This research was approved by the Animal Ethics Committee of the University of Tasmania (Permit # A0007591 and # A0008676), and by the Biodiversity Conservation Branch to take Wildlife (Permit # FA 03171 and # FA 05282) and Plants (Permit # FL 05036) for Scientific Purposes.

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Abstract

Fire management has become an increasingly critical issue in areas of high conservation value such as the pyrogenic buttongrass moorlands in the Tasmanian Wilderness World Heritage Area. The moorland avifauna is depauperate, comprised of only three cryptic, ground-dwelling resident species that depend exclusively upon moorlands in the study area. These include the Southern Emu-wren (*Stipiturus malachurus*), Striated Fieldwren (*Calamanthus fuliginosus*), and Ground Parrot (*Pezoporus wallicus*), in addition to a small number of species that are typically associated with adjacent forested habitats. This thesis is the first comprehensive study of the buttongrass moorland avifauna and investigated responses to post-fire succession primarily to help guide fire and conservation management. The replicated space-for-time study included sites in low productivity, blanket moorlands at Lake Pedder ($n = 12$; 2-54 years post-fire) and in moderate productivity, eastern moorlands at Lake St Clair ($n = 14$; 1-44 years post-fire). Avifaunal diversity, density, and habitat use over three seasons were quantified and analysed in relation to fire age, soil productivity and composition, structure, and spatial characteristics of habitats at both locations. Observed patterns of avifaunal diversity, density, and habitat use across the two chronosequences were complex and revealed high levels of inter-specific and inter-site variation in relation to habitat variables. Overall, mean densities of the resident species at Lake Pedder increased across the chronosequence, whereas at Lake St Clair they peaked 2-8 years post-fire. Mean densities of the non-resident species did not exhibit any consistent trends in relation to fire age. Observations of habitat use demonstrated that the resident and non-resident species used riparian and edge habitats disproportionately to their availability at both locations when compared to the moorland matrix. Surveys of potential arthropod prey resources conducted in matrix and riparian habitats at Lake St Clair indicated that mean abundance and mean energy content across orders were greater in riparian habitats and mid-seral sites, respectively. Thus, patterns of habitat selection by insectivorous species at Lake St Clair also appeared to reflect the differing availabilities of potential arthropod prey. Lastly, a paired before-after-control-impact study conducted at Lake St Clair ($n = 4$) indicated that hazard-reduction burning in moorlands may result in overall reductions in resident avian densities and increases in non-resident densities in the short-term (< 1.5 years post-fire). The implications of these findings are discussed in relation to current fire management practices and recommendations are provided to facilitate the conservation of critical resources for the moorland avifauna across the landscape and over time.

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Bumbling through Buttongrass

by Todd A. Chaudhry

Golden plains under whimsical skies
Hummocks 'n puddles and big march flies

Lurching leeches and yabbies abound
The Roaring Forties often the only sound

For the cryptic birds and sable snakes
Slink through the sedges and next to the lakes

Wombats amble through the tussock maze
On a bed of peat where it can rain for days

But a spell of sunshine can dry the mire
And a simple spark unleashes the fire

Burning buttongrass and tea-trees too
Enabling the moorland to grow anew

A world in miniature on the grandest of scales
Forever serenaded by the wailing westerly gales

Chapter 1

Introduction

Background

Fire is one of the primary abiotic agents in Australian ecosystems, as exemplified by the mosaic of vegetation types with contrasting fire response patterns that characterise much of the continent (Jackson 1968; Bowman 2000; Clark *et al.* 2002). Although recent evidence indicates that climatic factors initiated extensive landscape-scale changes in vegetation patterns, fire probably served to accelerate these trends (Kershaw *et al.* 2002). Characteristics of fire regimes (e.g. time since fire, season, patchiness, extent, intensity, frequency) influence the spatiotemporal patterns in plant communities, which affect faunal species composition and abundance through subsequent changes in the biotic and abiotic environment (e.g. habitat structure, food resources, microclimate) (Catling and Newsome 1981; Recher and Christensen 1981; Brown 1991; Whelan *et al.* 2002). Fire may cause direct effects on the fauna during and after single events (e.g. mortality, natality, emigration/immigration), and indirect cumulative effects due to habitat disturbance over time (e.g. population density, composition, persistence) (Fox 1978; Russell and Rowley 1993; Woinarski and Recher 1997; Whelan *et al.* 2002; Bradstock *et al.* 2005). The nature and extent of these effects are influenced by complex interactions with environmental factors such as landscape attributes (e.g. soils and topography) and pre- and post-fire climatic conditions (Keith *et al.* 2002a; Whelan *et al.* 2002). Faunal responses will also largely depend upon the specific life cycle attributes of the species of interest (Whelan *et al.* 2002), and may range from null responses for some generalist species (e.g. Kotliar *et al.* 2002), to temporary increases due to greater post-fire availability of food resources (e.g. Woinarski 1990), to reduction and recovery reflecting changes in habitat structure and suitability (e.g. Baker 2000). However, such patterns and their underlying processes are often very complex, and may only become apparent from long-term demographic studies (e.g. Brooker 1998).

Humans have used fire as a land management tool for millennia, varying from purposeful ignition and, more recently, to outright suppression (Pyne 1994; Jackson 1999a; Kershaw *et al.* 2002). Early Aboriginal changes in fire regimes during the Pleistocene have been implicated in the megafaunal extinction that took place in Australia between 50,000 to 45,000 years ago (Miller *et al.* 2005). Although fire activity is believed to have been relatively constant throughout the Holocene, there was a marked increase in anthropogenic burning during early European colonisation, which was followed by a reduction to current levels (Kershaw *et al.* 2002). Changes in fire regimes since European colonisation and the

resulting direct or indirect mortality have similarly either been confirmed or implicated as contributing to the extinction of at least two species and three subspecies of Australian birds (Woinarski and Recher 1997). Currently, fire regimes that are outside of their historical range of variation are a threat to at least 51 bird taxa, including many heathland species (Garnett 1992; Woinarski 1999a, 1999b; Garnett and Crowley 2002; Olsen and Weston 2005). Short inter-fire intervals and increased fire frequency are considered to be the major threats to many bird species, particularly for mid- to late-successional species that cannot persist or reproduce in early successional habitats (Brooker and Rowley 1991; Mushinsky and Gibson 1991; Woinarski and Recher 1997). Although many studies have been conducted on the effects of fire on Australian birds (for a review see Woinarski 1999a, 1999b), it is generally recognised that due to the extremely wide range of observed responses and complex interactions between a multitude of biotic and abiotic factors, ecosystem-specific research is essential in order to help guide prudent conservation and fire management activities (Wilson 1994; Woinarski and Recher 1997; Whelan *et al.* 2002).

Buttongrass moorlands form an ecosystem that exemplifies the complex interplay of fire, soils, flora, and avifauna. They are comprised of sedgeland and graminoid heathland communities typically dominated by the hummock-forming tussock sedge commonly named buttongrass (*Gymnoschoenus sphaerocephalus*) (Specht 1979a; Jarman *et al.* 1988a). Buttongrass moorlands are most extensive in the perhumid, oligotrophic peatlands of western Tasmania where they are largely protected within the Tasmanian Wilderness World Heritage Area (TWWHA) (Brown *et al.* 1993; Smith and Banks 1993). Buttongrass moorlands are recognised as a World Heritage ecosystem as they are highly pyrogenic, exemplify post-fire successional processes, occur in peatlands primarily formed by sedges as opposed to *Sphagnum* moss as in the Northern Hemisphere, and are largely undisturbed by development (Jarman *et al.* 1988a; Balmer *et al.* 2004). In part due to these unique characteristics, buttongrass moorlands are a difficult environment to live in and support a relatively depauperate fauna (Driessen 2006). In particular, the avian community consists of only three cryptic, ground-dwelling resident species that are thought to depend exclusively upon moorlands in the study area for survival and reproduction, namely the Southern Emu-wren (*Stipiturus malachurus*), Striated Fieldwren (*Calamanthus fuliginosus*), and Ground Parrot (*Pezoporus wallicus*). A small number of transient species that are typically associated with adjacent woodlands and related habitats are also present (Brown *et al.* 1993; Driessen 2006). Our knowledge of the moorland avifauna is very limited and primarily based on qualitative observations (e.g. Brown *et al.* 1993; Driessen 2006), since no detailed, community-level studies have been conducted to date. Bryant's study (1991) on the density, distribution, and conservation status of the Ground Parrot in Tasmania is the only significant research to date that has focused on a moorland resident species; however, its scope in

relation to fire ecology was limited and it did not investigate any other members of the avifauna (S. Bryant pers. comm. 2003). The rest of our knowledge of the resident species is limited to either old observational studies conducted in non-moorland habitat in other regions of the State (e.g. Legge 1908; Fletcher 1913a, 1913b, 1915a; Lord 1927; Sharland 1953), or more recent studies primarily on different subspecies on the Australian mainland (McFarland 1991a, 1991b, 1991c; Gosper and Baker 1997; Burbidge *et al.* 2005; Maguire 2006a, 2006b). Accordingly, over the years a number of researchers have identified the need to specifically study the Tasmanian moorland avifauna, particularly in relation to the effects of fire on the resident species, in order to help guide conservation efforts (e.g. Gellie 1980, Eberhard 1987; Bryant 1991).

Similar to many fire-adapted ecosystems around the world, there is considerable debate concerning the most appropriate way to manage fire within the TWWHA to conserve its biodiversity, particularly within buttongrass moorland ecosystems (DPIW 2007). The Parks and Wildlife Service (PWS; Department of Primary Industries, Parks, Water and Environment) is responsible for fire management within the TWWHA, and has mandates to both conserve natural and cultural resources, and to protect life and property (PWS 1999). The current strategy to meet these sometimes conflicting demands consists of limited, but fairly frequent hazard-reduction burning of buttongrass moorlands (i.e. inter-fire interval \leq 10 years) along areas that pose the greatest risk for ignitions (e.g. roads); and suppression, where and when it is feasible, throughout the rest of the landscape (Marsden-Smedley *et al.* 1999; PWS 1999). This fire regime has apparently caused changes in some of the plant and animal communities; however, very little is actually known about these patterns and processes. Nevertheless, some evidence suggests that current fire regimes represent a significant shift from historical regimes (Brown 1999; PWS 1999; Marsden-Smedley and Kirkpatrick 2000). Despite this relative lack of knowledge, there seems to be a growing recognition that the current fire management strategy may need to be changed to more closely mimic the historical disturbance regime characteristic of this ecosystem and to help conserve the diversity of species that comprise and depend upon it (PWS 1999; Marsden-Smedley and Kirkpatrick 2000; DPIW 2007). Accordingly, the Tasmanian Wilderness World Heritage Area Management Plan has identified fire research as one of its Key Focus Areas (PWS 1999). As part of this ongoing effort, the PWS and the Biodiversity Conservation Branch (BCB; Department of Primary Industries, Parks, Water and Environment) initiated a series of integrated research programs investigating the role of fire with respect to fauna, flora, and soils in buttongrass moorlands. This program, focused on assessing the impacts of fire on fauna, is a World Heritage Area Consultative Committee Priority One Project; however, to date it has been limited to studies on invertebrates and small mammals (e.g. Driessen 1999; Driessen and Greenslade 2004). Hence, in a recent

report for the Committee, one of the research needs that was specifically identified was to conduct a space-for-time project studying the effects of fire on birds in buttongrass moorlands (Driessen 2001). This thesis directly addresses that research need and will hopefully aid in the development of a better understanding of avian community ecology in buttongrass moorlands, provide an indication of overall ecosystem health (Mac Nally *et al.* 2004), and serve as a foundation for the implementation of sound conservation practices by fire and biodiversity managers. In addition, since this study has been developed and implemented in close collaboration with the BCB and PWS, the collective results from our research will be instrumental in developing a more holistic understanding of apparent patterns in avian habitat relationships and the underlying ecological processes in buttongrass moorlands. Furthermore, since some of the birds found in moorlands are migratory and/or are closely related to similar taxa on the Australian mainland, the possible applications of this research may reach well beyond the borders of the TWWHA itself. This is of particular significance since some of these taxa are either threatened in other parts of Tasmania, closely related to taxa that are threatened on the mainland, or otherwise potentially susceptible to altered fire regimes (see below).

In this thesis I used an integrated approach towards studying post-fire habitat relationships of the bird community in buttongrass moorlands by incorporating a range of disciplines including ornithology, fire ecology, vegetation classification, geographic information systems (GIS), entomology, dendrochronology, and pedology. Most of these disciplines, as they pertain to buttongrass moorland ecosystems, have been discussed to varying degrees in the literature. I have briefly described these below to provide a general framework for the thesis and I discuss them in more detail in subsequent chapters. However, I have provided a comprehensive literature review on the buttongrass moorland avifauna with an emphasis on the three resident species that are the focus of this thesis, since no such reviews have been conducted to date. I conclude this chapter by providing a description of the overall aims and structure of my thesis.

Buttongrass moorlands

Description

Buttongrass moorlands are extensive vegetation communities dominated to varying degrees by shrub and graminoid species, and most notably by the hummock-forming tussock sedge buttongrass (*Gymnoschoenus sphaerocephalus*) (Balmer 1991). Buttongrass has sclerophyllous leaves that form tussocks up to 2 x 2 m atop large rhizomatous pedestals (Brown 1999) (Figure1).



Fig. 1. Buttongrass (*Gymnoschoenus sphaerocephalus*)
(Photo: M. Driessen).

Buttongrass moorlands are defined in the standard reference by Jarman *et al.* (1988a) as being any treeless vegetation that typically contains buttongrass, or any vegetation in which buttongrass is common, but may contain widely spaced emergent trees. Small recurring islands (i.e. copses) or strips of vegetation along drainages (i.e. riparian zones) can be found within this main moorland matrix that may not contain buttongrass and are otherwise structurally and/or floristically distinct from the surrounding matrix (Jarman *et al.* 1988a; Marsden-Smedley 1990). Excluding such emergent features, buttongrass moorland vegetation is typically less than two metres in height (Balmer 1991). Moorlands are considered to be intermediary between terrestrial and wetland systems since they are waterlogged most of the year (Harris and Kitchener 2005).

Distribution

Buttongrass moorlands occur in isolated patches on mainland Australia including southeast Queensland, New South Wales, Victoria, and South Australia, but are most extensive and diverse in Tasmania (Keith 1995; Balmer *et al.* 2004). Buttongrass moorlands cover about 8% (0.55 million ha) of Tasmania and can be found throughout the State in a wide range of climatic, topographical, geological, and edaphic conditions (Jarman *et al.* 1988a; Balmer *et al.* 2004; TVMP 2004). However, they are most widespread in the perhumid, oligotrophic peatlands of lowland and subalpine western and southwestern Tasmania where they have likely dominated the landscape throughout the Holocene (Macphail 1979; Brown *et al.* 1993; Tye 2002; Bridle *et al.* 2003; Fletcher and Thomas 2007). They cover approximately 24% (335,000 ha) of the TWWHA (Figure 2), and they form part of a complex mosaic of vegetation consisting of wet scrub, wet sclerophyll forest, and cool temperate rainforest (TVMP 2004; Harris and Kitchener 2005).

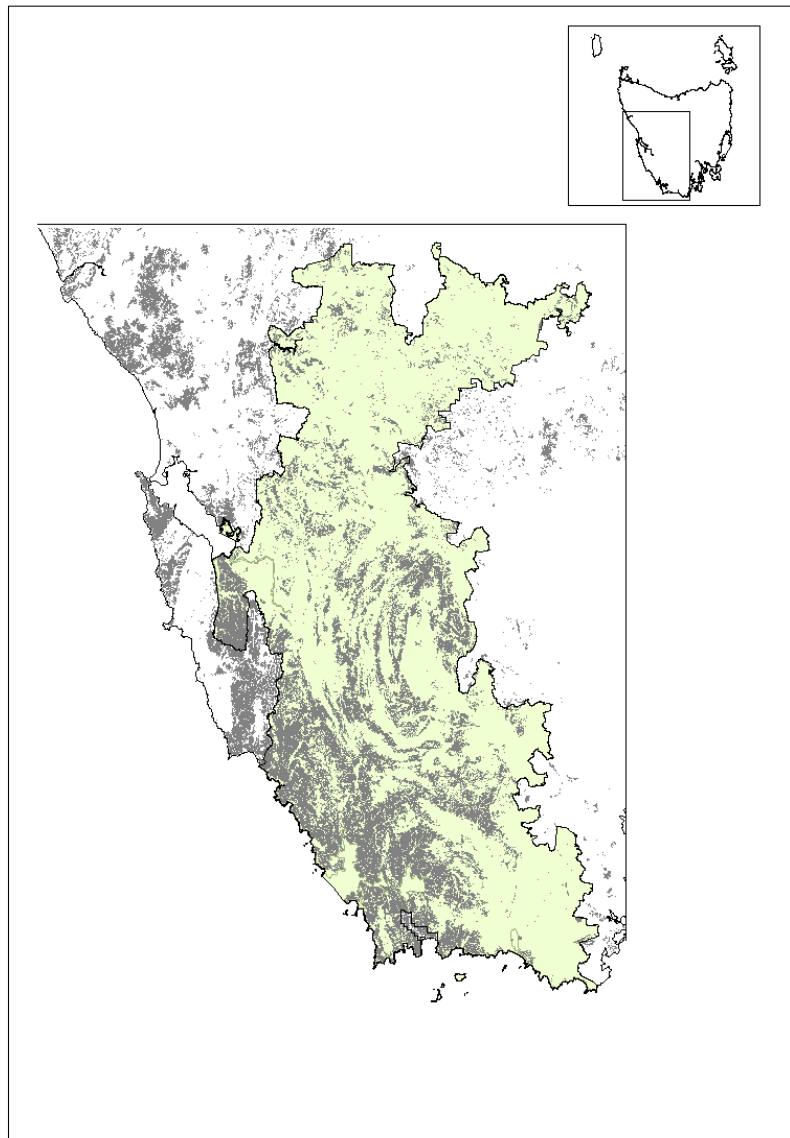


Fig. 2. Distribution of buttongrass moorlands and associated vegetation communities ■ in the Tasmanian Wilderness World Heritage Area ■ , Tasmania (Harris and Kitchener 2004; TVMMP 2009).

Floristics and structure

Buttongrass moorlands contain more than 206 vascular species in 50 families, with a relatively high level of Tasmanian endemism (Jarman *et al.* 1988a; Balmer *et al.* 2004). Most of the flora consists of graminoids (39%) from the Cyperaceae, Restionaceae, and Poaceae families; shrubs (32%) from the Epacridaceae, Myrtaceae, and Proteaceae; forbs (25%) from the Apiaceae; and ferns and their allies (4%) (Harris and Kitchener 2004; see Chapter 3). Many of the species that are found within these families are sclerophyllous, evergreen, and long-lived perennials that exhibit some adaptations to frequent fire, nutrient stress, and waterlogging (Jarman *et al.* 1988a; Brown 1999; Harris and Kitchener 2004; Balmer *et al.* 2004). These characteristics are common to species comprising similar heathland and related habitats in other regions of Australia and the world (Specht 1979a; Jarman *et al.* 1988a). In Tasmania, this diversity of species can form a floristically and

structurally variable mosaic of sedgeland, heathland, graminoid heathland, and wet scrub communities (Jarman and Crowden 1978; Jarman *et al.* 1982; Jarman *et al.* 1988a; Marsden-Smedley 1990).

Classification

Jarman *et al.* (1988a) utilised these differences in floristic composition and structure, along with climate, geography, and topography to develop a comprehensive classification system for buttongrass moorlands. Over the years buttongrass moorlands have been referred to by a range of names, such as hummock sedgelands (Jackson 1968), heathland/sedgelands (Jarman and Crowden 1978); sedgeland-heaths (Brown and Podger 1982) and wet temperate heaths (Keith *et al.* 2002a), but will be referred to as buttongrass moorlands throughout this thesis. Buttongrass moorlands share floristic and structural similarities with other heathlands and related communities that are typically (co-)dominated by woody plants less than two metres tall, such as those found in mainland Australia and other regions of the world, including New Zealand, South Africa, and northwestern Europe (Specht 1979a). The two major groups of buttongrass moorland identified by Jarman *et al.* include blanket moorland and eastern moorland, both of which have some lowland (~ 0 -600 m asl) and highland ($\sim > 600$ m asl) forms (Figures 3 and 4; see Chapter 3). Although floristic composition is one of the primary variables used to classify these groups, they share approximately 40% of their species in common (Harris and Kitchener 2004). Blanket moorlands occur extensively in western and notably in southwestern Tasmania, including the Lake Pedder region. They are primarily found on peat soils overlying infertile siliceous substrates (e.g. quartzite) from sea level up to approximately 1000 m asl and form a 'blanket' over a range of landscape features including flats, slopes, plateaus, and ridges (Jarman *et al.* 1988a; Harris and Kitchener 2005). Blanket moorlands consist of 15 main communities, the majority of which are graminoid heathlands (as defined by Specht 1979a), and a few peripheral communities and special habitats (Jarman *et al.* 1988a). Their boundaries with adjacent vegetation communities (e.g. forests) are often indistinct, creating ecotones in which floristics and structure intergrade (Jarman and Crowden 1978). Eastern moorlands are less widespread and occur primarily in isolated patches in eastern and central Tasmania. They are most extensive on poorly-drained peat flats and gentle slopes overlying moderately fertile substrates (e.g. dolerite) above 600 m asl on the Central Plateau, including the Lake St Clair region (Harris and Kitchener 2005). Eastern moorlands consist of 10 main communities, the majority being sedgelands that often contain scattered heath species, and a few peripheral communities and special habitats (Jarman *et al.* 1988a). Unlike blanket moors, eastern moors form relatively distinct boundaries with adjacent woodland communities that typically occur on the better-drained mineral soils associated with glacial moraines (Jarman *et al.* 1988a). Such minor hydrologic and edaphic differences may also facilitate the development of habitat features such as

copses and riparian vegetation zones within the primary moorland matrix (Jarman *et al.* 1982; Marsden-Smedley 1990). These factors are also at work in determining the nature of the transitions between the primary plant communities, which may vary from indistinct ecotones to distinct habitat edges (Jarman *et al.* 1982; Bowman *et al.* 1986; Balmer 1990), depending on the scale and degree of changes in these abiotic and biotic variables. These habitat edges and features may provide important microhabitats for birds, such as perching sites in emergent vegetation and eucalypts (Balmer 1990). Repeated use of such sites over time by birds and other vertebrates and the associated accumulation of excreta may also lead to increases in local nutrient levels, and thus help to perpetuate these communities (Verbeek 1984; Balmer 1990; Hannan 1993).

Fire ecology of buttongrass moorlands

Historical fire regimes

Fire has been part of the Tasmanian environment for approximately 30 million years (Kirkpatrick *et al.* 1978). Archaeological evidence indicates that Aborigines occupied southwestern Tasmania, although not necessarily continually, from approximately 35,000 years ago (Kee *et al.* 1993). However, recent palaeoecological research suggests that an increase in fire frequency from approximately 40,000-70,000 years ago may possibly be attributable to anthropogenic activities (Jackson 1999a; Kershaw *et al.* 2002), since this period did not coincide with major climatic changes and large, lightning-caused fires have been relatively infrequent in Tasmania until recent years (Jackson and Bowman 1982; Marsden-Smedley 1998a; Kershaw *et al.* 2002; J. Marsden-Smedley pers. comm. 2007). The following description of historical fire regimes in the study area is based on a comprehensive review and assessment by Marsden-Smedley (1998a). Historical records indicate that Tasmanian Aborigines actively burnt buttongrass moorlands throughout the study regions at the time of early European settlement in the early 1800s, although their actual time of occupation and initial use of fire as a management tool is debatable. They probably used fire to aid in hunting and gathering of specific resources, creating and maintaining travel corridors, communication, and warfare (Plomley 1966; Gammage 2008). Marsden-Smedley speculated that Aborigines probably lit frequent fires (e.g. inter-fire interval ≤ 20 years) under weather conditions that would result in low intensity burns that were largely restricted to moorland vegetation. Such conditions are typically associated with spring and autumn, but may also occur during dry periods in winter or after rain events in summer. Aborigines probably employed such a fire regime until their extirpation during the 1830s, and their activities may have facilitated the maintenance and expansion of buttongrass moorlands throughout western Tasmania prior to European colonisation (Jackson 1968;



Fig. 3. Blanket moorland, Lake Pedder region, Tasmania.



Fig. 4. Eastern moorland, Lake St Clair region, Tasmania.

Marsden-Smedley 1998a). The phase of early European settlement from the 1830s to the 1930s was characterised by periods of few, relatively small fires, which caused fuels to accumulate to levels that eventually led to high-intensity, landscape-scale fires. This regime largely reflected shifting patterns of European resource utilisation. Of particular note are fires during 1897/98 and 1933/44 that burnt large portions of the study area, including both moorland and fire-sensitive vegetation such as rainforest and alpine communities (Marsden-Smedley 1998b; Johnson and Marsden-Smedley 2002). Since the 1930s, fire has been prevented or suppressed throughout most of the region except for about a dozen large fires, and some habitat management and fuel reduction burns, particularly during the 1970s and more regularly in recent years. These fires were of variable intensity and the majority of the area burnt over this period was in moorland and wet scrub (Marsden-Smedley 1998b; Marsden-Smedley and Kirkpatrick 2000). As of 2007, it was estimated that approximately 65% of buttongrass moorland in the TWWHA had succeeded to old-growth (> 35 years post-fire), while only 23% was in a mature (15-35 years post-fire) and 12% in a regrowth seral stage (< 15 years post-fire), due to this considerable reduction in the frequency, intensity, and extent of fires (Marsden-Smedley 2007).

Post-fire succession

Although buttongrass moorlands have no direct commercial value, they have been the focus of a considerable amount of research and management over the years due to their extreme flammability and unique flora, fauna, and soils (e.g. Jackson 1968; Mount 1979; Bowman *et al.* 1986; Jarman *et al.* 1988a; Marsden-Smedley 1998a; Balmer and Barnes 2000; Brown *et al.* 2002; Bridle *et al.* 2003). Buttongrass moorlands are classified as having very high flammability and they will readily burn throughout the year, except after recent precipitation, but they have low fire sensitivity since many of the species are highly adapted to fire (Pyrke and Marsden-Smedley 2005). Most commonly crown fires scorch or burn through the entire above ground fuel array and typically have a moderate to high rate of spread and intensity, while surface fires burn fuels directly on the ground and typically have a low to moderate rate of spread and intensity (Gellie 1980; Jarman *et al.* 1988a; Marsden-Smedley 1993). Most moorland species regenerate quickly after fire, typically by vegetative regeneration through rootstock, but also by seed germination for some species (Jarman *et al.* 1988a; Balmer 1991; Brown 1999). Since under typical conditions the majority of above-ground vegetation is completely burnt, the age of most of the vegetation in a site will reflect the date of the last major fire event (Bowman and Jackson 1981; Jarman *et al.* 1988b). However, if prescribed burns are conducted under marginal conditions then some patches may remain unburnt (e.g. riparian zones) and a significant amount of thatch (i.e. near-surface dead fuels) may remain and pose a major fire hazard from several days up to approximately two years after the burn (Gellie 1980; Marsden-Smedley 1993; Marsden-Smedley and Catchpole

1995a, PWS unpublished data). Although rare, under extremely dry conditions buttongrass tussocks and other plants may be killed and peat fires may ignite (Marsden-Smedley 1993). Peat fires are characterised by little or no flaming combustion below the ground surface and very slow rates of spread, but are extremely difficult to extinguish (Gellie 1980; Marsden-Smedley 1993). Post-fire recovery of moorland vegetation may depend on a range of factors such as species composition, age, drainage, internal competition, external invasion, soil fertility, and post-fire weather conditions (Gellie 1980; Jarman *et al.* 1982; Bowman *et al.* 1986).

Despite the fact that successional processes remain poorly understood (Balmer *et al.* 2004), time since fire, fire frequency, and fire behaviour are considered to be the primary determinants of both floristics and structure in many buttongrass moorlands (Jarman and Crowden 1978; Brown and Podger 1982; Jarman *et al.* 1988a; Brown *et al.* 2002; Harris and Kitchener 2005). The most widely accepted model developed to help explain the vegetation patterns that characterise western Tasmania is 'ecological drift', as originally proposed by Jackson (1968) (Brown and Podger 1982; Jarman *et al.* 1988a; Brown *et al.* 2002; Balmer *et al.* 2004). This model is based on the concept that buttongrass moorlands are the first sere in a successional pathway; in the absence of disturbance by fire moorlands gradually change into wet scrub, then into wet sclerophyll forest, and ultimately climax as temperate rainforest (Jackson 1968). Despite the fact that the environmental conditions in perhumid western Tasmania appear to be suitable for the development of rainforest (e.g. precipitation > 1200 mm yr⁻¹), both historical and current patterns in vegetation communities show a disproportionate area covered by disclimax communities (i.e. buttongrass moorlands) (Jackson 1968, 1999a; Gammage 2008). Thus, Jackson (1968) proposed that fire frequency was the primary determinant of these vegetation patterns. Specifically, he stated that shorter inter-fire intervals favoured the development and abundance of pyrogenic communities across the landscape, and that longer inter-fire intervals favoured the development of fire-sensitive communities. Since each seral stage differs in its relative flammability, fire sensitivity, and productivity, each sere has a characteristic frequency distribution for fire-free intervals and thus, along with other environmental factors (e.g. topography and prevailing wind direction), helps to perpetuate the observed vegetation mosaic (Jackson 1968; Marsden-Smedley and Kirkpatrick 2000; Brown *et al.* 2002; Pyke and Marsden-Smedley 2005). Based on quantitative ageing of scrub and tree stems, Jackson suggested the average fire-free intervals required to maintain the different communities were approximately 20-40 years for buttongrass moorland, 40-80 years for wet scrub, 80-150 years for wet sclerophyll forest, and 150-300 years for rainforest (Jackson 1968, 1999a). Based on these estimates and paleoecological data, Jackson (1968, 1981, 1999a) proposed that relatively frequent burning by Aborigines throughout western Tasmania over the course of up to 70,000 years may have

resulted in buttongrass moorland extending beyond its edaphic limits (i.e. a vegetation disclimax), whereas in the absence of such frequent burning most of the region would be covered in rainforest.

A number of studies have been conducted over the years to investigate the validity of Jackson's model. Consistent with Jackson's model, the majority of moorlands (95%) studied by Jarman *et al.* (1988b) supported vegetation less than 50 years old and anecdotal evidence suggests that after approximately 65 years post-fire significant structural and floristic changes may occur (Marsden-Smedley 2003). Brown and Podger's (1982) study on a vegetation sequence in the Southwest also provided general support for ecological drift, but they suggested that there are numerous successional pathways that may also depend upon the duration and intensity of fire. Henderson and Wilkin's (1975) modelling based on Jackson's postulates accurately predicted the expected distribution of vegetation communities, with early successional communities dominating the landscape. However, a number of researchers have highlighted the importance of edaphic factors on vegetation dynamics. For example, Brown and Podger (1982) found that if conditions result in the combustion of peat, a single fire can change community composition and structure and may result in the successional process being slowed or halted (Brown and Podger 1982). Bowman *et al.* (1986) likewise suggested that the rate and extent of post-fire recovery is dependent on the fertility of the remaining peat layer, and thus the time scales proposed by Jackson may only be valid at sites with well developed and fertile peats. Brown *et al.*'s (2002) study that examined vegetation change over a 20 year period in the Southwest found that there were significant changes in both moorland structure and floristics, as well as shifts in vegetation boundaries between moorland and scrub, that appeared to be influenced by the time since the last fire. However, at some sites the effects of fire age were confounded with infection by the plant pathogen *Phytophthora cinnamomi* (see below). Although there were clear successional changes within the moorlands studied by Brown *et al.* (2002) (J. Balmer pers comm. 2003), a period of 20 years was apparently not long enough to observe marked changes from moorland to scrub, and they suggested that in areas with relatively infertile soils moorland communities could be maintained with longer inter-fire intervals. In addition, they suggested that although fire frequency affected the average species response to disturbance over time, the extent and distribution of vegetation communities appeared to be more a result of being topographically exposed to or protected from fire than fire frequency *per se*. Furthermore, it has been conjectured by Pemberton (1986, 1989) and Jarman *et al.* (1988a, 1988b) that edaphic factors such as waterlogging and severe frosts may result in conditions that are only conducive to supporting moorlands in some areas, such as the Central Plateau. In such areas moorlands may be able to persist even in the absence of fire, and thus do not conform to the ecological drift model.

In summary, as with other communities, vegetation cover, height, and structural complexity of moorlands are initially reduced by fire disturbance, and may then gradually increase to pre-fire levels through regeneration and reproduction (Brown 1991; Driessen 1999; Brown *et al.* 2002). The rate of post-fire recovery may be influenced by edaphic factors, and the intervals between fires may influence which species and hence communities are able to survive and perpetuate themselves (Marsden-Smedley 1990). Despite the ongoing debate regarding the exact nature of post-fire changes in moorland vegetation, for the purposes of this thesis these changes will generally be referred to as succession. This is in recognition of the possibility that such changes in the moorlands that comprise the chronosequences investigated in this thesis may be more due to changes in structure than in floristics *per se*, as implied by the classic models of plant succession (Noble and Slatyer 1981; Jarman *et al.* 1982; Brown 1991; Begon *et al.* 1996; Brown *et al.* 2002).

Fire behaviour and fuel modelling

Fire in buttongrass moorlands often exhibits more extreme behaviours (e.g. higher intensities and rates of spread) than would be expected under similar conditions in other vegetation communities. This is in part due to their high proportion of fine and especially dead fuels, open nature that allows rapid drying, high levels of volatile oils, and heterogeneous fuel characteristics (Gellie 1980; Balmer 1991; Marsden-Smedley 1993; Marsden-Smedley and Catchpole 1995b, 1995c). In fact, moorlands can burn at higher fuel moisture levels than any other communities for which data have been reported (Marsden-Smedley and Catchpole 1995b, 1995c; Marsden-Smedley *et al.* 1999; Balmer *et al.* 2004). Accordingly, they can sustain fires after only one or two rain/dewfall-free days and while in standing water (Marsden-Smedley and Catchpole 1995c; Marsden-Smedley *et al.* 1999).

Given the importance of fire in buttongrass moorlands and recognising that existing fire models did a poor job of predicting moorland fire behaviour, cooperative research on fuel and fire dynamics was conducted since the early 1990s in order to develop a specific fire behaviour prediction system for Tasmanian buttongrass moorlands (Marsden-Smedley *et al.* 1999). Research was focused on blanket moorlands, including sites at Lake Pedder, and sedgey eastern moorlands, including sites at Lake St Clair. The former were chosen since the majority of moorlands are classified as such, and the latter since significant fire management problems occur in these communities (e.g. high fire spread rates and flame heights) (Marsden-Smedley and Catchpole 1995b). Marsden-Smedley and Catchpole's results indicated that total and dead fuel loading could be reasonably predicted based on both geology and vegetation age (i.e. time since last fire). Moorlands were categorised as either low or moderate productivity sites, which is presumed to be a result of their associated geologies, including Precambrian quartzite at Lake Pedder and Jurassic dolerite at Lake St

Clair. Fuel loading was found to be positively correlated with site age. Fuel loading at the low productivity sites, ranging in age from 3-41 years post-fire, increased steeply at first and then stabilised beyond approximately 20 years. Fuel loading at medium productivity sites, ranging in age from 1-20 years, increased even more rapidly, as expected. However, since older moderate productivity sites were not available to sample, they could not confirm whether fuel loading likewise stabilises beyond approximately 20 years, although they considered such a trend to be ecologically reasonable. The primary factors that influenced fire behaviour, including rate of spread and flame heights, were wind speed, vegetation age, dead fuel moisture, and site productivity. Dead fuel moisture was, in turn, influenced by recent and significant rain/dewfall, temperature, and humidity (Marsden-Smedley and Catchpole 2001). Although sufficient data were not available, it is likely that slope has a similar influence on moorland fire behaviour as in other vegetation communities (Marsden-Smedley and Catchpole 1995b). The fuel and fire behaviour models that were the product of these studies are currently being used by PWS to guide prescribed burning and wildfire control operations in buttongrass moorlands, as outlined below.

Current fire management

Buttongrass moorlands are the focus of fire management activities in the TWWHA and adjacent lands since they are highly pyrogenic, can burn throughout the year, and because moorland fires can pose a risk to resources such as peat soils, fire sensitive vegetation communities, life, and property (Hannan 1993; Marsden-Smedley *et al.* 2001). Although *Sphagnum* bogs, rainforest, alpine, and plantation communities are classified as low to moderately flammable, they can burn under dry conditions, are highly to extremely sensitive to fire, and may take hundreds of years to recover (TVMP 2004; Pyrke and Marsden-Smedley 2005). The likelihood of moorland fires threatening such adjacent resources is purported to increase with higher fuel loads that are largely determined by the time since the last fire and site productivity (Balmer 1991; Marsden-Smedley and Catchpole 1995b; Marsden-Smedley and Kirkpatrick 2000). Therefore, fires in older moorlands have higher rates of spread and intensities that may significantly limit effective fire control, and thus result in resource damage or loss and enable the development of severe, landscape-scale wildfires (Marsden-Smedley and Catchpole 1995b, 1995c; Marsden-Smedley *et al.* 1999; Marsden-Smedley *et al.* 2001). Accordingly, most fire management activities in buttongrass moorlands are focused on resource protection, and consist of wildfire suppression and frequent tactical hazard-reduction burning in high-risk areas (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley *et al.* 2001). The primary objective of hazard-reduction burning is to reduce > 70% of the fuel load across > 70% of the site being burnt, which is typically contained by both natural (e.g. watercourses, wet scrub, forest) and constructed boundaries (e.g. roads and fuel breaks) (PWS 1996; Marsden-Smedley *et al.* 1999, Marsden-

Smedley *et al.* 2001; Marsden-Smedley 2009). Hazard-reduction burns have been conducted on a 5-8 year rotation at medium productivity sites at a high risk of accidental and arson ignitions, such as along the Lyell Highway at Lake St Clair (PWS 1996; Marsden-Smedley *et al.* 1999; Marsden-Smedley and Kirkpatrick 2000; see Chapter 2). However, it is recognised that such high frequency fire regimes may adversely affect biodiversity and cause long-term community changes (Jackson 1978; Marsden-Smedley and Kirkpatrick 2000; Pyrke and Marsden-Smedley 2005). Currently, it is estimated that less than 2% of moorlands throughout Tasmania is subjected to tactical hazard-reduction burning (Driessen 2006); however, recent fire modelling suggests that increased levels of burning (i.e. 5-10%) may be required to reduce the incidence and extent of unplanned fires in southwestern Tasmania (King *et al.* 2006, 2008). Only very limited habitat-management burns have been conducted to maintain suitable foraging habitat for the migratory Orange-bellied Parrot at Birchs Inlet and Melaleuca (Marsden-Smedley *et al.* 2001; J. Marsden-Smedley pers. comm. 2003).

Habitat loss and degradation

Since the majority of the study area is within the TWWHA it has been largely protected from anthropogenic disturbances other than fire. However, before its protection the study area was dramatically affected by hydroelectric power generation schemes that created large impoundments, including Lake Pedder (1972) and Lake Gordon (1974) in the Southwest and Lake King William (1951) in the Central Plateau (Hydro Tasmania 2007). It has been estimated that the total area of flat moorland vegetation inundated was 191 km² and 117 km² at Lakes Pedder and Gordon, respectively (Driessen *et al.* 2006). Other vegetation communities that may have served as suitable habitat for the avifauna were also inundated (e.g. wet scrub) at Lake Pedder (Balmer and Corbett 2001). This is probably the case at Lake Gordon as well, and thus these figures may be conservative estimates of total habitat loss. Although no similar assessment of pre-flooding vegetation has been conducted for Lake King William, the total area currently inundated is approximately 44 km²; based on topography and current vegetation it is probable that a sizeable proportion of this area was also composed of suitable moorland and associated communities (LIST 2003; TVMP 2004). While no studies have been conducted to date, the large loss of habitat due to inundation has probably had an impact on both avian species abundance and metapopulation dynamics in the study area, particularly for species such as the resident Southern Emu-wren that are known to have limited flight capabilities and perceive large expanses of unsuitable habitats (e.g. lakes) as barriers to dispersal (Littlely and Cutten 1994; Pickett 2000; Wilson and Paton 2004).

The other potentially significant disturbance to buttongrass moorlands is the introduced water mould, *Phytophthora cinnamomi* (Schahinger *et al.* 2003). It is a soil-borne plant pathogen that moorland communities are highly susceptible to and it has caused significant changes in moorland floristics and structure across extensive areas in Tasmania, notably in the Southwest (Brown *et al.* 2002; Schahinger *et al.* 2003). These effects may potentially be confounded or exacerbated by those of different fire regimes (Podger 1990; Brown *et al.* 2002). Despite the recognition that *P. cinnamomi* presents a significant threat to moorlands in the TWWHA, no studies have examined the potential impacts on the moorland avifauna (see Chapter 7).

The avifauna of buttongrass moorlands- a review

Despite the fact that buttongrass moorlands cover approximately 8% of Tasmania (Balmer *et al.* 2004; TVMP 2004), they support a relatively depauperate avifauna that largely consists of widely distributed and opportunistic species, similar to the avifaunas of other heathlands and related habitats found in mainland Australia, as well as other regions of the world such as South Africa, Europe, and North and South America (Cody 1975; Bigalke 1979; Gimingham *et al.* 1979; Kikkawa *et al.* 1979; Recher 1981; Specht 1994; Brown *et al.* 1993; Wirtz *et al.* 1996; Keith *et al.* 2002a). This may be due to an overall lack of adequate food resources in buttongrass moorlands, comparable to other cool and wet habitats in Tasmania (Ridpath and Moreau 1966), or due to their relative structural uniformity (Brown *et al.* 1993). The depauperate state of the buttongrass moorland avifauna is discussed in relation to other comparable communities and associated environmental factors in Chapter 7.

To date, no quantitative, comprehensive, and community-level avian surveys have been conducted in Tasmanian buttongrass moorlands. Wilson (1950), Ridpath and Moreau (1966), Rose (1978), Gellie (1980), Collins (1990), Brown *et al.* (1993), and Driessen (2006) all provide some description of the avifauna found in buttongrass moorlands and associated habitats (e.g. scrub). Overall, their descriptions were not based on formal surveys, did not include clear habitat definitions, and did not specify criteria for species inclusion. Accordingly, they should be interpreted with some caution and may not accurately reflect the avifauna of the areas covered by this study (see Chapter 2). They reported a total of 55 species utilising buttongrass moorlands and associated habitats to varying degrees (Table 1). A total of 15 species are common to at least five of the sources. According to Brown *et al.* (1993), which is the most thorough and detailed source, the bird community of buttongrass moorlands consists of approximately 18 species. For comparison, a total of 120 terrestrial bird species have distributions within the TWWHA as a whole (Driessen and Mallick 2003). Although the reported species assemblages may vary based on season, geography, and the precise definitions of habitat (P. Brown pers. comm. 2005), these figures provide a rough

estimate of the relatively low species richness that is typical of buttongrass moorlands. All sources agree that only three species appear to be common residents that depend exclusively on moorlands for breeding, feeding, and other resource needs within the study area. These specialist species are the Southern Emu-wren (*Stipiturus malachurus*), Striated Fieldwren (*Calamanthus fuliginosus*), and Ground Parrot (*Pezoporus wallicus*), and are the primary focus of this study. In addition, the Orange-bellied Parrot (*Neophema chrysogaster*) is dependent upon buttongrass moorlands for foraging in its geographically restricted summer breeding grounds in southwestern Tasmania (Brown and Wilson 1982). Although the remainder of the species listed in Table 1 have been recorded using buttongrass moorlands or associated habitats, none of them breed or permanently reside in the habitat. In this sense, these species are only secondary or marginal members of the buttongrass moorland community (i.e. non-residents of moorlands) and are either habitat generalists or more commonly associated with ecotonal and adjacent habitats (e.g. scrub and woodland).

Relatively few detailed studies have been conducted on the resident species of buttongrass moorlands in Tasmania or on the Australian mainland (refer to Higgins 1999; Higgins *et al.* 2001; Higgins and Peter 2002). Accordingly, some of the information presented below is from studies conducted within other regions of each subspecies' respective ranges or on other subspecies. Whenever possible, information most relevant to this study and consistent with personal observations has been presented. Since the non-resident species are not the primary focus of this study they are not described in detail below, but references to them are provided throughout this thesis as appropriate.

Table 1. Avifauna recorded in Tasmanian buttongrass moorlands and associated habitats. Table adapted from Brown *et al.* (1993) and taxonomy follows Christidis and Boles (1994). * Moorland resident species, ^ Tasmanian endemic species, + Introduced species. *A- Habitat* (as reported, therefore some categories overlap): 1. Dependent on moorlands; 2. Opportunistic feeders, open ground especially after fire; 3. Shallow surface water; 4. Raptors/aerial hunters; 5. Near taller creekside vegetation (1-5 per Brown *et al.* 1993; Gellie 1980); 6. General- Wet tussock sedeglands and moors (Ridpath and Moreau 1966), buttongrass moorlands or scrub (Wilson 1950; Gellie 1980; Driessen 2006), heath/sedgeland and scrub (Rose 1978), heath and scrub (Collins 1990). *B- Status* in buttongrass moorlands (Ridpath and Moreau 1966; Gellie 1980; Sharland 1981; Collins 1990; Brown *et al.* 1993; Simpson and Day 1999; Watts 2002; P. Brown pers. comm. 2005): C = Common, O = Occasional, U = Uncommon, VR = Very Rare, N = Not specified; R = Resident (in Tasmania), M = Migratory. *C- Diet* (Sharland 1981; Brown *et al.* 1993; Watts 2002; P. Brown pers. comm. 2005): S = granivore, I = Insectivore, O = Omnivore, H = Herbivore, C = Carnivore. *D- Source* (of species inclusion): B = Brown *et al.* 1993; C = Collins 1990; D = Driessen 2006; G = Gellie 1980; R = Rose 1978; RM = Ridpath and Moreau 1966; W = Wilson 1950.

Common name	Scientific name	Habitat ^A	Status ^B	Diet ^C	Source ^D
Southern Emu-wren *	<i>Stipiturus malachurus</i>	1	C-R	I	B, D, RM, G, R, C
Striated Fieldwren *	<i>Calamanthus fuliginosus</i>	1	C-R	I	B, D, RM, G, W, R, C
Ground Parrot *	<i>Pezoporus wallicus</i>	1	C-R	S	B, D, RM, G, R, C
Orange-bellied Parrot	<i>Neophema chrysogaster</i>	1	VR-M	S	B, D, G, C
Black Currawong ^	<i>Strepera fuliginosa</i>	2	O-R	O	B, D, W, R, C
Forest Raven	<i>Corvus tasmanicus</i>	2	O-R	O	B, D, W, R, C
Goldfinch +	<i>Carduelis carduelis</i>	2	O-R	S	G
Grey Shrike-thrush	<i>Colluricincla harmonica</i>	2	O-R	I,C	G, R, C
Richard's Pipit	<i>Anthus novaeseelandiae</i>	2	O-R	I,S	B, D, RM, W
Brown Goshawk	<i>Accipiter fasciatus</i>	2,4	U-R	C	G
Grey Goshawk	<i>Accipiter novaehollandiae</i>	2,4	U-R	C	G, R, C
Dusky Robin ^	<i>Melanodryas vittata</i>	2,5	C-R	I	B, D, G, R, C
Scarlet Robin	<i>Petroica multicolor</i>	2,5	O-R	I	G
Sulphur-crested Cockatoo	<i>Cacatua galerita</i>	2	O-R	H,S	B, D, R, C
Latham's Snipe	<i>Gallinago hardwickii</i>	3	O-M	I	B, D, G, R, C
Lewin's Rail	<i>Rallus pectoralis</i>	3	U-R	I	B
Brown Falcon	<i>Falco berigora</i>	4	C-R	C	B, D, W, R, C
Collared Sparrowhawk	<i>Accipiter cirrhocephalus</i>	4	N-R	C-I	R, C
Southern Boobook	<i>Ninox noaveseelandiae</i>	4	N-R	C, I	R, C
Swamp Harrier	<i>Circus approximans</i>	4	O-M	C	B, D
Tree Martin	<i>Hirundo nigricans</i>	4	C-M	I	B, D, W, R, C
Wedge-tailed Eagle	<i>Aquila audax</i>	4	U-R	C	RM, W, R, C
Welcome Swallow	<i>Hirundo neoxena</i>	4	C-M	I	B, D, RM, R, C
White-bellied Sea-eagle	<i>Haliaeetus leucogaster</i>	4	U-R	C	RM
Beautiful Firetail	<i>Stagonopleura bella</i>	5	C-R	S	B, D, G, R, C
Blue-winged Parrot	<i>Neophema chrysostoma</i>	5	O-M	S	W
Brown Thornbill	<i>Acanthiza pusilla</i>	5	VR-R	I	W
Fan-tailed Cuckoo	<i>Cacomantis flabelliformis</i>	5	C-M	I	B, RM, R, C, BP
Flame Robin	<i>Petroica phoenicia</i>	5	O-M	I	W, R, C
New Holland Honeyeater	<i>Phylidonyris novaehollandiae</i>	5	C-R	I,H	D, W, R, C
Olive Whistler	<i>Pachycephala olivacea</i>	5	O-R	I	W, R, C
Pallid Cuckoo	<i>Cuculus pallidus</i>	5	U-M	I	RM

Table 1. cont.

Common name	Scientific name	Habitat ^A	Status ^B	Diet ^C	Source ^D
Superb Fairy-wren	<i>Malurus cyaneus</i>	5	O-R	I	D, RM, W, R, C
Yellow-throated Honeyeater ^	<i>Lichenostomus flavicollis</i>	5	C-R	I	B, D, W, R, C
Silvereye	<i>Zosterops lateralis</i>	5	C-M	I,H	W, R, C
Striated Pardalote	<i>Pardalotus striatus</i>	5	U-M	I	W
Tasmanian Thornbill ^	<i>Acanthiza ewingii</i>	5	C-R	I	D, W, R, C
Yellow Wattlebird ^	<i>Anthochaera paradoxa</i>	5	U-R	I, H	W, R, C
Bassian Thrush	<i>Zoothera lunulata</i>	6	N-R	I, S	R, C
Black-faced Cuckoo-shrike	<i>Coracina novaehollandiae</i>	6	N-M	I	R, C
Brown Quail	<i>Coturnix ypsilophora</i>	6	U-R	S,I	RM
Crescent Honeyeater	<i>Phylidonyris pyrrhoptera</i>	6	C-M	I,H	D, G, W, R, C
Eastern Spinebill	<i>Acanthorhynchus tenuirostris</i>	6	N-R	H, I	R, C
Golden Whistler	<i>Pachycephala pectoralis</i>	6	N-R	I	R, C
Great Egret	<i>Ardea alba</i>	6	N-M	C, I	C
Green Rosella	<i>Platycercus calydonicus</i>	6	N-R	S	R, C
Grey Fantail	<i>Rhipidura fuliginosa</i>	6	N-M	I	R, C
Masked Lapwing	<i>Vanellus miles</i>	6	U-R	I	RM
Pink Robin	<i>Petroica rodinogaster</i>	6	N-R	I	R, C
Scrubtit ^	<i>Acanthornus magnus</i>	6	N-R	I	R, C
Strong-billed Honeyeater ^	<i>Melithreptus validirostris</i>	6	N-R	I	R, C
Tasmanian Scrubwren ^	<i>Sericornis humilis</i>	6	N-R	I	R, C
White-throated Needletail	<i>Hirundapus caudacutus</i>	6	N-M	I	R, C
Yellow-tailed Black-cockatoo	<i>Calyptorhynchus funereus</i>	6	N-R	S,I	R, C

Southern Emu-wren

Description

The Southern Emu-wren is a small passerine with an approximate length of 14-19 cm, wingspan of 9-19 cm, and weight of 5-9 g (with the Tasmanian subspecies being at the lower end of these ranges) (Higgins *et al.* 2001). Overall its plumage is rufous above and streaked blackish, and orange-buff below. Notably, it has a long filamentous tail that resembles Emu feathers (Figure 5). There is no seasonal variation in plumage and sexes are fairly similar except for the prominent sky-blue supercilium and bib found on juvenile and adult males (Higgins *et al.* 2001).



Fig. 5. Male Southern Emu-wren (Photo: Courtesy of M. Pickett).

Taxonomy and distribution

The Southern Emu-wren (*Stipiturus malachurus*) comprises seven (Rowley and Russell 1997) to eight (Schodde and Mason 1999) subspecies in the family Maluridae and is endemic to, and patchily distributed throughout, eastern and southern Australia (Schodde 1982). The Tasmanian subspecies, *Stipiturus malachurus littleri*, is distributed through the eastern, and locally common throughout the northern and western regions of the State, particularly within the TWWHA (Thomas 1979; Schodde 1982; Schodde and Mason 1999; Driessen and Mallick 2003). Although a number of studies have been published on the Southern Emu-wren, some of the following aspects of its biology and ecology are not generally known (Higgins *et al.* 2001).

Habitat

Southern Emu-wrens utilise a diversity of habitats throughout their range that typically consist of low, dense vegetation (Higgins *et al.* 2001). Specific habitats include wet and dry heaths and associated ecotones, sedgelands, buttongrass moorlands, coastal dunes, and

wetland areas such as bogs, fens, swampy gullies, and reedlands (Fletcher 1915a; Sharland 1981; McFarland 1988b; Pickett 2000). Although their habitat can be structurally diverse, it is typically dense, ranging from 0.5-1.5 m tall, and has few trees (Wilson and Paton 2004). Some research indicates that such structural parameters may be more important than floristics in determining habitat suitability (i.e. able to support survival and reproduction) (Pickett 2000; Wilson and Paton 2004; Maguire 2006a). Although in parts of their range they are more commonly found in undisturbed natural habitats, they have been reported using woodland regrowth 2-3 years post-logging (Wardell-Johnson and Williams 2000) and sometimes utilise introduced vegetation (e.g. blackberry thickets) found in more developed areas, particularly in autumn and winter in Tasmania (McNamara 1937). They have been recorded from sea level to approximately 1000 m asl (Schodde 1982).

Populations

Southern Emu-wrens can be locally common in suitable habitat, but only a few studies have provided estimates of home range sizes and densities (Sharland 1981). Fletcher (1915a) estimated that one pair's territory consisted of approximately 1 km of swampy creekside habitat in Tasmania. Recent quantitative research conducted by Maguire and Mulder (2004) in Victoria showed that pairs defended territories with a mean of 0.97 ± 0.09 ha (range 0.30-2.86 ha); however, territory boundaries were more variable during the winter. Pickett (2000) reported similar estimates from the Mount Lofty Ranges of South Australia, with breeding season mates having highly overlapping home ranges with a mean of 0.85 ± 0.66 ha (range 0.34-2.61 ha), and non-breeding individual home ranges of 0.31-6.53 ha. Southern Emu-wren densities have been estimated by Gosper and Baker (1997) who reported a minimum density of 1.6 birds ha⁻¹ in dry and wet heathlands in New South Wales, Maguire (2006b) who reported a mean density of 2.3 birds ha⁻¹ in wetlands and coastal heathlands in Victoria, as well as by McFarland (1988b) and Jordan (1987a) (see below).

Behaviour

Southern Emu-wrens are cryptic, ground-dwelling birds and can be very elusive and difficult to study (Pickett 2000). This is particularly true under windy conditions, in which they tend to remain hidden in low cover (Rowley and Russell 1997). Maguire and Mulder (2004) reported that birds were only visible for 6.6% of the total of 1434 person-hours of surveying they conducted. They are sedentary residents that can be found individually, but more often in pairs or small family groups, and can exhibit strong territoriality through interspecific physical contests and song displays (Gosper and Baker 1997; Maguire 2006b). They creep and run along the ground with 'mouse-like' movements and when disturbed they often seem to prefer to hop away into denser cover where they can easily conceal themselves (North 1912; Fletcher 1913a; Schodde 1982). However, males in particular may sometimes move

to a more prominent position in order to investigate the disturbance before retreating (Sharland 1981; Pickett 2000). They are considered to be weak and reluctant fliers (Pringle 1982a), and when approached within close range they typically only flush up to 30 m, after which they may be difficult to relocate (Corben 1973; Schodde 1982). Most individuals seem to perceive large open areas that do not provide adequate cover (e.g. paddocks and lakes) as barriers to dispersal (Littlely and Cutten 1994; Pickett 2000; Wilson and Paton 2004), but may disperse through otherwise unsuitable habitat (e.g. mature forest) (Wardell-Johnson and Williams 2000). Juveniles have been recorded dispersing up to 1.2 km (Maguire and Mulder 2004), and the longest recorded one-way movements of banded Southern Emu-wrens were approximately 2.5 km (Pickett 2000).

Calls

Southern Emu-wren territorial songs are high-pitched (~ 5-12 kHz) and variable descending trills composed of four to six rapid *deedle* notes (Pringle 1982a; Rowley and Russell 1997; Higgins *et al.* 2001). Their songs, often issued from prominent perches, can be heard throughout the year (Rowley and Russell 1997; Pickett 2000). Although both sexes have been observed singing, it is most often done by males and during the breeding season (Fletcher 1915a; Rowley and Russell 1997). Southern Emu-wrens also frequently issue soft *tsuuuh* contact calls while foraging, and a buzzy *trrrt* alarm call when disturbed (Schodde 1982; Rowley and Russell 1997). Although they are often very difficult to hear, especially when there is noise disturbance such as strong winds (Pickett 2000), they can be heard up to approximately 50 m under good listening conditions (Schodde 1982).

Diet

Southern Emu-wrens are primarily insectivorous and only rarely consume plant material (Barker and Vestjens 1989). They typically glean, and occasionally sally for invertebrates, as they hop along the ground and particularly up and through shrubs (Fletcher 1915a; Littlely and Cutten 1994; Rowley and Russell 1997). They have been observed foraging in both open and dense vegetation, but appear to favour the latter, particularly when alarmed (Littlely and Cutten 1994; Wilson and Paton 2004). They have also been observed picking insects from spider webs, and preying on large insects during courtship feeding and the provisioning of nestlings (Gosper and Baker 1997; Maguire and Mulder 2004; Maguire 2006b). They prey on species from a wide range of Insecta and Arachnida orders including but not limited to the following: Araneae, Coleoptera, Diptera, Hemiptera, Hymenoptera, Lepidoptera, Mantodea, Neuroptera, and Orthoptera (Lea and Gray 1936; Barker and Vestjens 1989).

Breeding

Southern Emu-wrens have been recorded breeding from August to January in Tasmania (Fletcher 1913a; Fletcher 1918). They make domed nests with side entrances, loosely constructed from vegetation such as grass and moss, and lined with feathers and other soft materials (Fletcher 1915a; Sharland 1981). The nests are often well concealed and placed in a range of locations, from low dense vegetation to recently burnt areas, and from ground level to the top of shrub patches (Fletcher 1915a). Southern Emu-wrens can have multiple broods with two to four (although usually three) eggs each (Fletcher 1913a; Sharland 1981). Young of the first brood are usually driven away before the second brood, the young of which may then stay with their parents as late as mid-winter (Fletcher 1915a; pers. obs.). Although considered uncommon, cooperative breeding by Southern Emu-wren males was verified by Maguire and Mulder (2004). Breeding success is variable across their range, and some populations have sustained significant losses due to high mortality and may be vulnerable to local extinction (Fletcher 1915a; Jordan 1987a; Pickett 2000; Maguire and Mulder 2004).

Fire ecology

Direct threats to Southern Emu-wrens as a result of fire include adult mortality, loss of clutches, and partial or complete loss of populations (Fletcher 1913a; Fox 1978; Pickett 2005). Their limited flight capabilities may render them particularly vulnerable to extensive wildfires (Cooper 1974; Pringle 1982a). However, a few accounts indicated that they are able to actively avoid fire fronts (McNamara 1945; Gellie 1980; Schodde 1982). Although fires during spring may interrupt breeding cycles, they have been observed surviving and re-nesting after some fires, and are able to temporarily colonise habitats regenerating after disturbance (Fletcher 1913a; Gellie 1980; Emison *et al.* 1987; Britton 2004). Small unburnt or partially burnt patches of vegetation, such as along creeks, may be necessary for them to persist in the post-fire environment (Recher and Christensen 1981; Pickett 2005). Gellie (1980) stated that if adequate unburnt vegetation is not available, then breeding in Tasmanian moorlands could be precluded 5-7 years post-fire; however, he did not provide any substantiating data. Jordan (1987a) observed Southern Emu-wrens foraging along the edges of recently burnt areas of Barren Grounds Nature Reserve, New South Wales, and presumably recolonising from adjacent source populations within a year after the fire. However, populations did not increase rapidly until a year after the fire and were most abundant (~ 40 birds 10 ha^{-1}) 2-3 years post-fire. The fluctuations observed in this population coincided with seasonal changes in the post-fire insect populations. Loyn (1997) similarly found that Southern Emu-wrens were common in their preferred heathy understorey habitats before an extensive wildfire in eastern Victoria, but their numbers declined steeply and remained at low levels until populations started to recover a couple

years post-fire. After a severe wildfire in southwestern Victoria, Reilly (1991a) observed some Southern Emu-wrens occasionally using a one hectare patch of heath that had burnt a year prior. However, they did not return to the larger burnt heath/woodland and swamp thicket site, and another heath/woodland site, until approximately 3 and 4.5 years post-fire, respectively. McFarland (1988b) reported densities of 0.8 birds 10 ha⁻¹ at 2.5 years post-fire, 1.2 birds 10 ha⁻¹ at 5.5 years, and 0.4 birds 10 ha⁻¹ at 6.5 years, while none were reported in the 0 and 10.5 year old heathland sites in Cooloolo National Park, Queensland. McFarland (1994) also reported that their highest densities of 2.0-2.5 birds 10 ha⁻¹ were at 6-8 year old sites, and no nests were found in heathlands < 2 and > 10.5 years post-fire. Such apparent delays in recolonisation, as noted above, may be partly attributed to Southern Emu-wrens need for adequate vegetative cover for foraging and shelter (Gellie 1980; Schodde 1982; MacHunter *et al.* 2009). High fire frequency from both wildfires and management burns, extensive wildfires in fragmented habitat, and otherwise modified fire regimes have been implicated in the local extinction of some populations, and have been identified as some of the primary threats to extant populations (Pringle 1982a; MLRSERT 1998; Garnett 1992; McFarland 1994; Maguire and Mulder 2004; Pickett 2005).

Threats and status

In addition to altered fire regimes, clearing, draining, and fragmentation of habitat have been identified as some of the primary threats to extant populations and have already led to declines and local extinctions in some areas (Schodde 1982; Rowley and Russell 1997). Although they can persist in small, isolated patches of habitat in some parts of their range (Schodde 1982), small subpopulations (e.g. < 30) are more prone to sudden population declines (Maguire and Mulder 2004). Habitat fragmentation in conjunction with altered fire regimes are likely to have contributed to the local extinction of some populations, as well as reduced the likelihood of recolonisation from potential isolated source populations (Garnett 1992; Littlely and Cutten 1994; McFarland 1994). Inbreeding suppression has also been considered as a possible threat to small, isolated populations in parts of their range (Pickett 2000). Southern Emu-wren broods in Tasmania have been parasitised by Horsfield's Bronze and Fan-tailed Cuckoos, and potentially by Pallid Cuckoos as well (Fletcher 1915b). Other threats include native predators such as rats, birds, and particularly snakes, introduced foxes and cats, and invasion of habitat by both native and introduced weeds (Pringle 1982; Maguire and Mulder 2004).

The Southern Emu-wren showed no significant change (< 20%) in national reporting rates between *The Atlas of Australian Birds* (1977-81; Blakers *et al.* 1984) and *The New Atlas of Australian Birds* (1998-2002; Barrett *et al.* 2003), but has decreased (> 20%) in portions of its range on the Australian mainland (Barrett *et al.* 2003). Accordingly, the Southern Emu-

wren has five mainland subspecies that are listed as being threatened, including: *S. m. intermedius* endangered in the Mount Lofty Ranges, *S. m. parimeda* endangered on the Eyre Peninsula; *S. m. halmaturinus* rare on Kangaroo Island; *S. m. malachurus* rare in the southeast; all of which are listed under the South Australia *National Parks and Wildlife Act 1972*. *S. m. intermedius* is also listed as endangered under the *Environment Protection and Biodiversity Conservation Act 1999* and critically endangered under the *Action Plan for Australian Birds 2000* (Garnett and Crowley 2000). In addition, *S. m. hartogi* is listed as rare or likely to become extinct on Dirk Hartog Island under the Western Australia *Wildlife Conservation Act of 1950*.

Striated Fieldwren

Description

The Striated Fieldwren is a small passerine with an approximate length of 14 cm, wingspan of 18 cm, and weight of 20 g (Higgins and Peter 2002). Overall its plumage is olive above, yellowish below, and heavily streaked blackish throughout. It has a pale chin and throat, a conspicuous supercilium, and a tail with a black subterminal band and whitish tip (Figure 6). There is no seasonal variation in plumage and sexes do not differ in size, but do differ slightly in plumage, with juveniles being very similar in plumage to that of their respective sex (Higgins and Peter 2002). However, given the similarity in appearances and their shy nature, particularly that of females (Chandler 1912; Sharland 1953), in practice it is very difficult to distinguish sexes in the field (pers. obs.).



Fig. 6. Striated Fieldwren (Photo: H. Stewart).

Taxonomy and distribution

The Striated Fieldwren (*Calamanthus fuliginosus*) comprises four subspecies in the family Pardalotidae, is endemic to southeastern Australia, and is considered to be most common in Tasmania (Pringle 1982b). The two Tasmanian subspecies only vary slightly in plumage and size and include the nominate *Calamanthus fuliginosus fuliginosus* distributed throughout the eastern half of the State, and *Calamanthus fuliginosus diemenensis*, distributed throughout the western half of the State (Thomas 1979; Schodde and Mason 1999). In the absence of additional data needed to determine intergradation of the two subspecies in central-south Tasmania, it is assumed that the latter subspecies is the focus of this study as the majority of sites fall in the wetter western watershed of Tasmania (Schodde and Mason 1999; Higgins and Peter 2002). Although a few studies have been published on the Striated Fieldwren, many of the following aspects of its biology and ecology are generally not known (Higgins and Peter 2002).

Habitat

Striated Fieldwrens utilise a diversity of habitats throughout their range that typically consist of low, dense vegetation (Higgins and Peter 2002). Specific habitats include dry and wet heathlands, sedgeland, tussock grasslands, alpine meadowlands, marshes, some dry and wet sclerophyll forests and associated clearings, and particularly buttongrass moorlands in Tasmania (Wilson 1950; Sharland 1953; Thomas 1979; Keast 1978; Gosper and Baker 1997; Taylor *et al.* 1997; Schodde and Mason 1999). In some areas they seem to prefer shrubbier sites with emergent shrubs (≤ 1 m) and may also utilise taller scrub edges (≤ 3 m) bordering such sites (Recher 1981; Gosper and Baker 1997). They can also be found along the borders of wetlands, such as marshes and creeks, and in introduced vegetation in more developed areas, such as along roads, fences, and ditches (Legge 1908; Napier 1969; Keast 1978; Sharland 1981). They have been recorded from sea level and occasionally up into subalpine forest (~ 900 - 1100 m asl) and above tree-line in Tasmania (> 1100 m asl) (Sharland 1953; Thomas 1979; MacDonald 2001).

Populations

Although their numbers can be hard to estimate (Watson 1955), some studies have reported approximate home range sizes and densities of Striated Fieldwrens. Taylor *et al.* (1997) estimated densities of 3.4 ± 1.1 birds 10 ha^{-1} in young regrowth (0-6 years old) and 0.6 ± 0.4 birds ha^{-1} in older regrowth (6-12 years old) and none in mature (> 12 years old) dry sclerophyll forest with heathy and sedgely understories in southeast Tasmania. Based on observations made in Victoria, Chandler (1912) estimated that home ranges were roughly 0.2-0.8 ha, and Keast (1978) observed that birds were spaced about 50 m apart where populations were most dense. Gosper and Baker (1997) found that they were uncommon

overall, but had a relatively high average minimum density of 6.2 birds 10 ha⁻¹ (range 4.0-9.0 birds 10ha⁻¹) in their preferred habitat of low, dry heath in New South Wales.

Behaviour

Striated Fieldwrens are considered to be shy and secretive ground-dwelling birds, and are not easily detected except by call (Legge 1908). Although they are resident and primarily sedentary, they have been recorded up to 160 km from the nearest known population in New South Wales (Higgins and Peter 2002). They are found singly, in pairs, and occasionally in small groups and exhibit territorial behavior throughout the year (Keast 1978; Gosper and Baker 1997). They typically run or creep along the ground and seem reluctant to fly unless approached closely (Sharland 1981; Pringle 1982b). After being disturbed they may quickly run away, drop into cover, or suddenly flush and fly up to approximately 20 m away (Littler 1904; Legge 1908; Sharland 1953; Keast 1978). They may be very difficult to relocate and flush again, or may perch on top of emergent shrubs appearing to scan the area for the source of the disturbance (Legge 1908; Chandler 1912; Gosper and Baker 1997). Despite their overall shy nature, it has been noted that they can be less wary in developed areas where their natural habitat has been modified (Legge 1908).

Calls

Striated Fieldwrens have complex, variable, and melodious territorial songs (~ 2-6 kHz) thought to be sung by males (Legge 1908; Chandler 1912; Sharland 1953; Stewart 2001; Higgins and Peters 2002). They typically sing from prominent perches, such as at the top of emergent shrubs, with their head tilted back and tail cocked and flicking from side to side (Legge 1908; Dove 1912; Keast 1978). They often start the song in a low key, seeming to come from hundreds of metres away, which has a ventriloquial effect (Chandler 1912; Sharland 1953). They then continue into a loud song that can be readily heard from up to approximately 100 m away and last for 20 seconds (Keast 1978; Gosper and Baker 1997). A common form begins with two sharp *whip*, *whip* notes, followed by a pause, a single note, another pause, and then 4-5 quick notes (Keast 1978). They will often cease singing if approached within approximately 30 m and then plunge into cover (Chandler 1912). They are known to sing at most times of the day and throughout the year, and particularly just before sunset during the early breeding season of winter and during spring mornings and afternoons (Sharland 1953; Watson 1955). In addition to their distinctive song, Striated Fieldwrens also have a couple of other notable calls, including a rapid trill and a short *churr* call, the latter typically issued during summer in Tasmania (Stewart 2001; T. Chaudhry unpublished data).

Diet

Striated Fieldwrens are primarily insectivorous, but do occasionally feed on seeds as well (Pringle 1982b). They have been observed spending approximately two-thirds of their time foraging in low or emergent shrubs, although they do not seem to prefer particular plant species, and one-third on the ground (Keast 1978; Gosper and Baker 1997). They prey on species from a wide range of Insecta, Arachnida, and Mollusca orders including but not limited to the following: Araneae, Coleoptera, Gastropoda, Hemiptera, Hymenoptera, Lepidoptera, Orthoptera, and Neuroptera (Lea and Gray 1936; Barker and Vestjens 1989).

Breeding

Striated Fieldwrens are one of the earliest breeders in Tasmania and have a long breeding season, beginning as early as July and sometimes extending into early February (Legge 1908; Dove 1912; Sharland 1953; Napier 1969). They construct dome-shaped nests with side or top entrances from vegetation such as dried grass, leaves, and moss, and line the interior of the nest with feathers or hair (Dove 1912; Sharland 1981). The nests are well concealed and located on or close to the ground and within or under tussocks or other low vegetation (Sharland 1981; Dove 1916). They have at least two broods a year with three to four eggs in each brood (Sharland 1981). Eggs have been found in nests from July to October (Fletcher 1915b), and fledglings from September to December in western Tasmania (Legge 1908; Dove 1912). Young have been observed in the nest as late as early February in eastern Tasmania (Napier 1969).

Fire ecology

Only limited and mostly qualitative information is available on the fire ecology of Striated Fieldwrens. Early records indicate the loss of natural habitat through ‘burning off’ (Legge 1908). Gellie (1980) stated that they are capable of finding sufficient food and cover almost immediately after fire, and have been observed foraging on recently burnt moorlands in Tasmania. He further stated that breeding cycles may be interrupted if fires occur during spring, and breeding may not be successful for five to seven years post-fire unless adequate unburnt patches are available; however, he provided no supporting data. Taylor *et al.*’s (1997) study indicated that the Striated Fieldwren occurred in significantly higher densities in young regrowth (0-6 years old) when compared to older regrowth coupes (6-12 years old) following clearfall logging, regeneration burns, and portions of both coupes being subsequently burnt in a wildfire; however, they were absent from mature forest (> 12 years old). In southwestern Australia, Striated Fieldwrens are considered to be an early successional species in heath, thus long-term fire exclusion may render some areas unsuitable (Smith 1985, 1987). It has been recommended that a mosaic of fire ages should

be provided for such heathland species in order to maintain current population levels (Woinarski 1999b).

Threats and status

Threats to the Striated Fieldwren, in addition to possible threats from fire, include extensive land clearing and drainage of wetlands for agricultural development (Napier 1969; Sharland 1981). Such habitat loss may have significantly reduced populations in these areas and possibly caused a shift in habitat utilisation toward some of these disturbed environments that have developed elements of suitable habitat (Legge 1908; Sharland 1981). Natural predators such as snakes have also been indicated as possible threats (Chandler 1912). In addition, the Fan-tailed Cuckoo and potentially the Horsefield's Bronze-Cuckoo and Pallid Cuckoo are brood parasites of the Striated Fieldwren in Tasmania (Fletcher 1915b; Brown *et al.* 1993).

The Striated Fieldwren showed no significant change (< 20%) in national reporting rates between *The Atlas of Australian Birds* (1977-81; Blakers *et al.* 1984) and *The New Atlas of Australian Birds* (1998-2002; Barrett *et al.* 2003), but decreased (> 20%) in western Tasmania and portions of its range on the Australian mainland (Barrett *et al.* 2003). Due to severe reductions in populations from some of the above threats (including broad-scale vegetation clearing, exotic weeds, and grazing pressure; DEWHA 2007) *C. f. albiloris* is listed as vulnerable under the New South Wales *Threatened Species Conservation Act 1995*. However, it is recognised that there is inadequate information available regarding the current status of this species (Lunney *et al.* 2000).

Ground Parrot

Description

The Ground Parrot is a psittaciforme and is one of only four ground-nesting parrots in the world (Balmer *et al.* 2004). It has an approximate length of 30 cm, wingspan of 13 cm, and weight of 80 g (Higgins 1999). Overall its plumage is bright green with black and yellow streaks throughout. It has a red band across the forehead and yellow wing bars (Figure 7). There is no seasonal variation and sexes of both adults and juveniles look similar, with juveniles lacking the red forehead band and having bolder black streaks (Higgins 1999).



Fig. 7. Ground Parrot (Photo: Courtesy of PWS).

Taxonomy and distribution

The Ground Parrot (*Pezoporus wallicus*) comprises two subspecies in the family Psittacidae, and is endemic in primarily coastal areas of eastern, southeastern, and western Australia. The nominate *Pezoporus wallicus wallicus* is found in eastern Australia, and Tasmania is considered to be its stronghold (Forshaw 1981). It is found throughout western and southeastern Tasmania, and has the greatest numbers in the Southwest, particularly within the TWWHA (Bryant 1991; Driessen and Mallick 2003). Unlike the Southern Emu-wren and Striated Fieldwren described above, the Ground Parrot has been extensively studied throughout its range (Higgins 1999).

Habitat

Ground Parrots primarily utilise patchily distributed heathland and sedgeland habitats within their range, with such areas typically lacking tall shrubs or trees (Forshaw 1981; Meredith 1983; McFarland 1988a, 1988c). In Tasmania, Bryant (1991) reported that they predominantly occur in graminoid heathlands (especially Standard and Layered Blanket Moors, after Jarman *et al.* 1988a) and sedgelands, and on well to poorly drained soils. The vegetation is typically 0.5 to 1.0 m tall (range 0.3-2.5 m). It is dense with a projective foliage cover > 60% from 0-30 cm above ground level, and a highly variable overall projective foliage cover of 30-90%. However, no significant correlation was found between Ground Parrot densities and projective foliage cover or plant species richness. The apparent importance of vegetation structure has also been noted in research conducted in New South Wales (Wall 1989). Within their home ranges, they likely use a mosaic of different habitat

types, such as sedgey areas for foraging and heathy areas for nesting and roosting (Bryant 1991). Such use of different microhabitats can change throughout the year and may be partly in response to seed abundance and accessibility (McFarland 1991a). Ground Parrots are found from sea level to alpine areas over 1300 m asl, in the approximately 570,000 ha of estimated potentially suitable habitat in Tasmania. However, most are found below 500 m asl in perhumid southwestern Tasmania (Bryant 1991).

Populations

In Tasmania, Bryant (1991) reported Ground Parrot populations ranged from 0.2-5.3 birds 10 ha⁻¹, with a mean density of 1.7 ± 1.0 10 ha⁻¹. The smallest inhabited patch of moorland was 5 ha. These data, in conjunction with the estimated area of potentially suitable habitat, were used to calculate a theoretical estimate of 97,000 birds. Some of her surveys were conducted in the areas covered by this study. Near Lake St Clair a mean of 2.1 birds 10 ha⁻¹ (range 1.5-3.1 birds 10 ha⁻¹) was reported for Burns Plains (3 years post-fire) and King William Plains (8 years years post-fire). None were reported along Lake St Clair Road. Near Lake Pedder, a mean of 1.2 birds 10 ha⁻¹ (range 0.3-2.1 birds 10 ha⁻¹) was reported for a number of sites along Scotts Peak Dam Road (11-19 years post-fire). Consecutive year surveys conducted at the latter sites indicated that densities remained fairly constant. Repeated surveys conducted at Snug Tiers near Hobart showed that estimated densities fluctuated throughout the year, with highest densities in March and November and lowest densities in June and July, corresponding with the breeding season, dispersal of juveniles, and mid-winter, respectively. No significant association was found between density and altitude (Bryant 1991). Extensive research on the Ground Parrot has also been conducted in Coolool National Park, Queensland by McFarland (1991a, 1991b, 1991c). Birds maintained an average distance of 106 m from each other. Mean home ranges were 9.2 ha (mean 5.6 ha for adults and 13.9 ha for subadults). There was considerable overlap of individual home ranges, but little in foraging areas (McFarland 1991a). These figures are comparable to Meredith *et al.*'s (1984) estimate that pairs require 8-20 ha of heath (depending on fire age). A study on Ground Parrot vocalisations also conducted in Queensland indicated that sites 5.4 km apart supported birds that were still considered to be from the same population or metapopulation (Chan and Mudie 2004).

Behaviour

Ground parrots are ground-dwelling and both diurnal and crepuscular (Jordan 1987b). Overall they are solitary and exhibit low intraspecific aggression (Courtney 1997); however, adults are territorial during the breeding season (Jordan 1987b). Due to their camouflaged plumage, preference for dense habitat, and shy and elusive nature, direct visual observations on the ground are very difficult (Forshaw 1981, McFarland 1991d). They are not easily

flushed even within close range (Burbidge 1989), and may simply remain motionless or quickly run away (McFarland 1991d). They are usually flushed singly, and only rarely in pairs (Bryant 1991). The mean bird-observer distance at flushing is 12.3 ± 0.5 m (range 2-50 m), and they typically fly in a zig-zag pattern into denser vegetation with a (straight-line) mean distance of 76.8 ± 3.3 m (range 4-200 m) (McFarland 1991d). If pursued, they usually only flush once or twice and then may resort to running away (Green and Mollison 1961; McFarland 1991d). Ground Parrots are considered to be strong fliers and are capable of flying long distances even over non-suitable habitat (e.g. woodland or water) (Forshaw 1981). By some accounts, they are considered to be primarily sedentary (Cooper 1974; Bryant 1994). However, radio-telemetry data showed that they can be fairly mobile, especially when young, with one bird moving a mean of 730 m over a 17 day period (Jordan 1987b). The longest recorded single flight was 700 m (McFarland 1991a) and others have been recorded dispersing up to 220 km from the nearest known breeding area (Meredith *et al.* 1984).

Calls

Ground Parrots are noted for having distinctive calling-flight sessions during dawn and dusk (Higgins 1999), and do occasionally call during the day, especially in spring and summer in some parts of their range (Jordan 1984; pers. obs.). Up to 11 calls types have been recognised; however, their typical call issued during their calling-flight sessions can be characterised as a high-pitched (~ 2 -5 kHz) and ascending flute-like whistle consisting of *tsee* notes (Watts 2002; Chan and Mudie 2004). The calls are issued by both sexes and most birds (79%) may call while in flight (Jordan 1987b; McFarland 1991b). It has been assumed, with some substantiating evidence, that all birds at a site call during each session, with the possible exception of birds dispersing after the breeding season (Meredith *et al.* 1984). Calls are sometimes issued simultaneously by different birds, but usually one call is followed by a response by another bird (Forshaw 1981). A number of studies have indicated that these calls can be heard for at least 400 m under still conditions with no rain, but only up to approximately 200 m under windy conditions with heavy rain (Meredith and Isles 1980; Bryant 1991). Light intensity is considered to be the primary influence on the beginning and end of calling sessions (McFarland 1991b). In Tasmania, they call for a mean of 20.3 ± 0.8 minutes at dusk, with a mean of 4.8 ± 0.2 calls bird⁻¹ and 0.3 ± 0.02 calls bird⁻¹ minute⁻¹ (Bryant 1991). These sessions were typically longer during the breeding season, and calling duration, number of calls, and rate of calling were all significantly correlated with density of Ground Parrots (Bryant 1991). They fly for approximately 10-15 minutes during the middle of the session, with a mean flight length of 220 m during dusk sessions (Jordan 1987b; McFarland 1991b). Although adults rarely fly more than 400 m, one young was recorded flying 1.74 km from its daytime foraging area to nighttime roost site (Jordan 1987b).

Diet

Ground Parrots are primarily granivorous, but may occasionally feed on other plant matter and insects (Sharland 1981). They usually feed on the ground, sometimes in low shrubs, and may consume the order of 10,000 seeds per day (Meredith and Isles 1980; Forshaw 1981; Jordan 1989). It has been estimated that males forage over an area of approximately 13 ha around the nest to provision chicks during the breeding season (Jordan 1984). They are considered to be opportunistic foragers, and likely choose seeds based on varying aspects of availability and size (McFarland 1991a). In Tasmania, their known and potential food plants come from 13 dicotyledon and 12 monocotyledon species from 9 families, with the majority of their food probably from the Restionaceae and Cyperaceae families (Bryant 1994). The wide variety of these species that can be found in graminoid heathlands and sedgeland may help to explain Ground Parrots' apparent preference for these habitats. However, as sites become older they may become more dominated by shrub species. The associated floristic and structural changes may result in a decrease in sedge seed abundance and accessibility (Jordan 1989).

Breeding

Ground Parrots have been recorded breeding from mid-October to February in Tasmania, but timing may vary depending on climatic conditions and food resources (McFarland 1988a; Bryant 1991). Their nests can be located in dry or ecotonal microhabitats (McFarland 1991b). They consist of shallow depressions in the ground lined with buttongrass stems, rushes or other plant material, and are well concealed under tussocks or small shrubs (Green and Mollison 1961; Hodges 1961; Sharland 1981). They usually only have one brood with a mean of 4.24 ± 0.12 eggs (range 2-6 eggs) from October to as late as January in Coolool National Park, Queensland (McFarland 1988a; McFarland 1991b). Egg failure rates can be high (22-31%), and pairs averaged 1.9 fledglings nest⁻¹ (Meredith and Isles 1980; McFarland 1991b). Chicks are fledged from December onwards, and young may remain in the natal area for a couple months before dispersing to new habitat (Meredith *et al.* 1984; Bryant 1991).

Fire ecology

Extensive research has been carried out on the fire ecology of the Ground Parrot throughout its range (Woinarski 1999a). Although studying the effects of fire on Ground Parrots was not considered to be the primary focus of her research in Tasmania (S. Bryant pers. comm. 2003), the majority of this section references Bryant (1991, 1992, 1994), with additional references from mainland research as appropriate.

Ground Parrots have been observed flying before fire fronts (Cooper 1975), but may potentially lose clutches if fires take place during the breeding season (Gellie 1980), and may die due to heat or asphyxiation during severe wildfires (Fox 1978). A Ground Parrot has also been observed foraging on grains in a fowl-yard after presumably being displaced from a nearby buttongrass plain by a bushfire (Fletcher 1946). Bryant (1991) identified Ground Parrots across the full range of sites investigated in Tasmania, from 1-90 years (mean 13.4) by flushing and from 2 months-35 years (mean 9.2) post-fire by call surveys. Recolonisation occurred at sites approximately 1 year post-fire, when the estimated minimal projective foliage cover of 30% was attained at most sites (Bryant 1994). However, Ground Parrots have been noted using sites much sooner after fires (Gellie 1980). Such younger sites, especially with small copses of older and denser vegetation, may provide adequate cover and become temporarily crowded within larger burnt areas (Meredith *et al.* 1984; Bryant 1991). Densities typically remained low until 4-7 years post-fire, when densities peaked above approximately 2.2 birds 10 ha^{-1} . Densities were then maintained above the mean of 1.7 birds 10 ha^{-1} until approximately 13 years post-fire, after which densities declined but remained good (density not specified) more than 30 years post-fire. The correlation between densities and fire age was highly significant ($\rho = 0.26$, $P < 0.001$, $n = 182$), and may be attributable to age related differences in flowering, seeding, and vegetation structure (Bryant 1991). Although fire frequency was not explicitly studied and no data were provided, she attributed some local extinctions to ‘overfiring’ of moorlands, specifically along the Lyell Highway corridor within the Lake St Clair study area (see Chapter 2).

Results from studies conducted on the mainland have shown similar correlations between Ground Parrot densities and site age. In Queensland, McFarland (1991c) reported Ground Parrots returned to subtropical heathland sites within one year post-fire, peaked at densities of 4-5 10 ha^{-1} at 5-8 years post-fire, while none were recorded at sites > 14 years post-fire. He attributed such long-term patterns to changes in vegetation structure and food resources, and short-term fluctuations with seasonal effects on population dynamics. In New South Wales, Jordan (1987b) similarly found that populations in a heathland peaked at 5-6 years post-fire, but declined to zero by 12 years post-fire. However, a long-term study conducted in the same area by Baker and Whelan (1994) and Whelan and Baker (1999) indicated that populations reached maximum densities 13 years post-fire, and ongoing monitoring has indicated that they are still present although apparently declining 20 years post-fire (Tasker and Baker 2005). In Western Australia, Western Ground Parrots have likewise been reported inhabiting older sites, and were considered to be relatively common in sites 20-40⁺ years post-fire, but can utilise recently burnt sites when directly adjoined to unburnt habitat with established populations (Burbidge 1989; Burbidge *et al.* 2007). Such an apparent range in the age of utilised habitats may reflect regional differences in rainfall and other factors,

which in turn affect the rate of vegetation regrowth. In this sense, vegetation structure (e.g. cover) and floristics (e.g. seed production) may have a greater influence on Ground Parrot densities than fire age *per se* (Meredith *et al.* 1984; Baker and Whelan 1994). Fire frequency has also been identified as an important consideration in managing Ground Parrot habitat (Meredith 1984a, 1984b; Bryant 1991; Burbidge *et al.* 2007). Short fire intervals (e.g. 6-8 years) may lead to declines and local extinctions due to a loss of plant species that Ground Parrots rely on for food. After being subjected to such fire regimes, some sites may take a long time to become suitable again (Meredith 1984a).

Clearly, fire plays a critical role in Ground Parrot densities, and some populations on the mainland have become threatened or locally extinct due to detrimental fire regimes, ranging from frequent prescribed burning to fire exclusion (Woinarski 1999b). However, fire is only one factor in a complex set of interactions, with other factors including vegetation type, habitat diversity, extent, and proximity to recolonisers (McFarland 1991c). Although Bryant (1991) stated that prescribed burning specifically for the Ground Parrot was not required, she provided some general guidelines in order to minimise adverse impacts and maintain adequate suitable habitat across the landscape. Guidelines included conducting prescribed burns between April and September in order to minimise impacts during the breeding season, providing a mosaic of burnt and unburnt vegetation within sites, and not burning more than 25% of an area over a 10 year rotation in order to maintain sustainable densities.

Threats and status

In addition to altered fire regimes, Ground Parrot populations are threatened by habitat loss, fragmentation and degradation from development, agriculture, grazing, and the plant pathogen *Phytophthora cinnamomi* (Forshaw 1981; Bryant 1991; Garnett 1992). Additional threats may include avian diseases and predation by both native (e.g. snakes and raptors) and feral animals (e.g. foxes and cats) (Lord 1927; Meredith 1984a; Jordan 1989; Bryant 1991). Despite numerous studies citing predation as a major threat to Ground Parrot populations, McFarland (1991c) considers the risks to Queensland populations to be minimal. In Tasmania, Ground Parrots populations have declined and some populations have become locally extinct, particularly in northern and eastern regions where suitable habitat was historically limited (Bryant 1991). Such losses have been attributed to development, grazing, and increased fire frequency. However, populations are considered to be secure in southwestern Tasmania where they are free from most threats under the protection of the Tasmanian *Nature Conservation Act 2002* and the TWWHA (Bryant 1991).

Due to some of the threats identified above, Ground Parrot populations and distribution have been severely reduced in parts of its range (Lunney *et al.* 2000). Accordingly, the Ground Parrot is listed as vulnerable under the *Action Plan for Australian Birds 2000* (Garnett and

Crowley 2000), the Queensland *Nature Conservation Act 1992*, and the New South Wales *Threatened Species Conservation Act 1995*. The Western Ground Parrot, *P. w. flaviventris*, is listed as endangered under the *Environment Protection and Biodiversity Conservation Act 1999* and the *Action Plan for Australian Birds 2000* and rare or likely to become extinct under the Western Australia *Wildlife Conservation Act of 1950*.

Orange-bellied Parrot

The Orange-bellied Parrot (*Neophema chrysogaster*) is extremely rare and within Tasmania is only found within the Macquarie and Bathurst Harbour basins in the Southwest. It is a summer migrant from mainland Australia that depends on buttongrass moorlands for foraging during the summer breeding season, and adjacent forests for nesting (Brown and Wilson 1984). Appropriate fire regimes are thought to be an important factor in maintaining optimal foraging and nesting habitat, but no rigorous fire ecology studies have been conducted to date (Brown *et al.* 1993; M. Holdsworth pers. comm. 2007). Due to significant habitat loss on the mainland and a very small population size, the Orange-bellied Parrot is listed as critically endangered under the *IUCN Red List of Threatened Species 2004*, and endangered under the Australia *Environment Protection and Biodiversity Protection Act 1999*, the Tasmania *Threatened Species Protection Act 1995*, and the South Australia *National Parks and Wildlife Act 1972*. However, since the Orange-bellied Parrot breeds outside of the range of this study (see Chapter 2), it is not considered any further herein.

Thesis aims and structure

This is the first quantitative, community-level study of the avifauna of Tasmanian buttongrass moorland and investigated the influence of a range of abiotic and biotic factors on avifaunal composition and densities, particularly in relation to post-fire succession. The primary aim of this thesis was to contribute information to help guide fire management and conservation of the moorland avifauna in the TWWHA, with a focus on the three resident species (i.e. Southern Emu-wren, Striated Fieldwren, Ground Parrot).

I wrote the primary Chapters (4-6) of this thesis in a format to facilitate publication in the near future. Accordingly, there is some repetition, but I have attempted to limit this by referring to other chapters, as appropriate. However, I have elaborated within some sections (e.g. Methods) to enable a more critical review of my research and the replication of methods for future research and monitoring. I am the principal contributor to all chapters in this thesis and will be the primary author for all publications and presentations arising from it.

In Chapter 2, I provide a rationale for the study design and site selection process, background information on the study locations and sites, and a critical evaluation of fire regime data for

the study sites. I rely upon this information for analyses and interpretation of results in subsequent chapters.

In Chapter 3, I present methods for developing a fine-scale and high-resolution classified avian habitat map for buttongrass moorlands and associated vegetation communities. I use this map to quantify the composition, structure, and spatial characteristics of avian habitat in the study area that are required for analyses and interpretation of results in subsequent chapters. In addition, I present the results from a formal accuracy assessment of the Tasmanian Wilderness World Heritage Area Vegetation Mapping (WHAveg) within the study area.

In Chapter 4, I use a replicated space-for-time (SFT) design to investigate the short- to long-term effects of fire (i.e. 1-54 years post-fire) and other environmental variables on avifaunal composition and densities in low productivity blanket moorlands at Lake Pedder and moderate productivity eastern moorlands at Lake St Clair, with an emphasis on the three resident species.

In Chapter 5, I investigate whether the resident and non-resident avian species use habitats disproportionately in relation to availability and fire age within low productivity blanket moorlands at Lake Pedder and moderate productivity eastern moorlands at Lake St Clair. I describe and compare the terrestrial arthropod community between moorland matrix and riparian habitats in relation to fire age, and investigate whether patterns of habitat use of insectivores reflect the patterns of potential availability of arthropod prey resources.

In Chapter 6, I opportunistically utilise prescribed burns conducted by Parks and Wildlife Service at two study sites included in the SFT study in Chapter 4. I use a before-after-control-impact design (BACI) to investigate the short-term responses of the moorland avifauna to hazard-reduction burning in relation to unburnt control sites.

In Chapter 7, I conclude this thesis by synthesising the effects of fire and other environmental factors on avian species composition, density, and habitat use in Tasmanian buttongrass moorlands. I discuss my results in relation to similarly depauperate sedgeland and heathland communities within a national and global context, as well as patterns in post-fire faunal succession within a theoretical context. I discuss the implications of results from previous chapters for fire and conservation management in Tasmanian moorlands, and conclude this thesis by providing recommendations for future research.

Chapter 2

Study design, site descriptions, and fire regimes

Study design

The ultimate aim of this thesis was to investigate the effects of post-fire succession on the avifauna of Tasmanian buttongrass moorlands in order to provide recommendations for their conservation to fire and wildlife managers. The literature is replete with studies investigating the effects of fire on Australian fauna, and on birds in particular (for a review see Woinarski 1999a, 1999b). However, since it is clear that reported patterns and their underlying processes are both complex and highly variable in space and time (Whelan *et al.* 2002; Burbidge 2003; Gill and Bradstock 2003), it is critical to conduct site- and species-specific research to develop appropriate management recommendations. There was a clear management need for this study as no previous studies have explicitly focused on the fire ecology of the Tasmanian moorland avifauna, and only one (Bryant 1992) has explored the effects of fire age on the resident Ground Parrot. Accordingly, the primary aims of this Chapter were: 1) to provide a rationale for the study and describe its design and the site selection process; 2) to present background information on the study locations and sites; and 3) to collect, compile, and evaluate data on fire regimes for the study sites.

Studies with comparable aims to this one have been conducted both on the Australian mainland and abroad. The inferences that can be drawn from many of these studies are limited by a host of issues associated with study design, implementation, analyses, and reporting. A number of different study designs are often used to assess the effects of fire on birds, including designed experiments, before-after-control-impact (BACI), space-for-time (SFT), and inferential studies (Stewart-Oaten *et al.* 1986; Pickett 1989; Loyn 1999; Block *et al.* 2001), each of which has its pros and cons. Replicated manipulative field experiments are considered to be the most desirable design since they make it possible to draw strong inferences regarding responses to fire regime parameters *per se* and to develop a more process-based understanding of population changes (Whelan *et al.* 2002). BACI studies that include randomly allocated replicate burnt and control sites and adequate pre- and post-fire surveying can likewise provide a basis for strong inferences (Whelan 1995; see Chapter 6). Despite their obvious advantages, studies that employ such rigorous designs are exceedingly rare (e.g. Loyn *et al.* 2003; Woinarski 1990) due to real-world constraints such as limited time, personnel, and funding. Furthermore, it is often not feasible to impose the treatment (i.e. fire) for designed experiments at the appropriate time, location, and scale (Whelan 1995). Thus, many studies must use sites that were either burnt in the past (e.g. Meredith *et*

al. 1984; McFarland 1988b) or opportunistically utilise previously surveyed sites subjected to contemporaneous fire events and accept the associated compromises in study design and applicability (e.g. Brooker and Rowley 1991; Recher 1997; Loyn 1997).

A SFT design was deemed the most appropriate and feasible option to utilise, considering the primary aims and specific constraints imposed on this study. SFT designs are commonly used within ecosystems that exhibit marked successional dynamics in relation to disturbance events, such as buttongrass moorlands, and are based on the assumption that spatial and temporal variations are homologous (Pickett 1989; Brown *et al.* 2002). In other words, many researchers are interested in identifying temporal trends over the medium to long term (i.e. ~ 10 to > 50 years) but do not have the ability to monitor sites over commensurate time scales. For that reason, sites of different ages (i.e. time since the last fire) are ‘substituted’ for time to create a chronosequence of sites from which retrospective temporal trends can be extrapolated (Pickett 1989). In this sense, SFT studies fall within the general class of comparative mensurative experiments, as defined by Hurlbert (1984). In a comprehensive review and annotated bibliography of Australian birds and fire by Woinarski (1999a, 1999b), he noted that a large proportion of studies used SFT or similar designs. Perhaps the most serious inherent limitation of SFT studies is the possibility of inter-site variability in abiotic (e.g. geology, topography, climate) and biotic (e.g. vegetation, disease, predation) factors confounding or compounding observed differences in populations from effects of fire regimes *per se* (Loyn 1999; Woinarski 1999b). Another limitation is that both population and fire regime differences may covary with these and other factors that may be either unmeasured or unknown (Whelan 1995). Specific deficiencies that have been identified for previous avian SFT studies in particular, and fire ecology studies in general, include failures to:

- incorporate adequate (or any) replication, control sites, survey timespans, and statistical analyses (Whelan 1995; Burbidge 2003);
- survey an adequate range of fire ages (Baker and Whelan 1994);
- clearly report and assess the quality of fire regime data and report the sizes and specific locations of fires, sites, replicates, and sampling units (Parr and Chown 2003);
- investigate the impacts of fire regime parameters other than just time since last fire on populations (Woinarski 1999b);
- quantify and investigate the impacts of other habitat variables (e.g. vegetation, productivity, food resources) (Smith 2000);
- and form explicit collaborations between researchers and land management authorities to facilitate the research process and implementation of management recommendations (Whelan 1995).

Despite some of these limitations and deficiencies, SFT and other mensurative studies are still deemed to be valuable and pragmatic approaches to assess post-fire changes in faunal communities, provided they are carefully designed, implemented, and interpreted (Twigg *et al.* 1989; Whelan 1995; Loyn 1999). Furthermore, they are viewed as a necessary prerequisite for much needed experimental research into ecosystems, such as buttongrass moorlands, for which such baseline information does not currently exist (Saab and Powell 2005).

To address several of the above design deficiencies, this SFT study included two study locations stratified by productivity and vegetation type, and comprising a broad chronosequence of replicated fire ages. To the extent possible, sites were selected to minimise inter-site variability in abiotic and biotic factors which have been described and quantified in detail. This study was designed and implemented in close collaboration with the Biodiversity Conservation Branch (BCB) and the Parks and Wildlife Service (PWS) (Department of Primary Industries, Parks, Water and Environment), which are responsible for fauna, flora, and fire management within the Tasmanian Wilderness World Heritage Area (TWWHA). This multidisciplinary collaboration included a synchronous SFT study conducted by BCB on the effects of fire on moorland invertebrates and was inclusive of all sites used in this study except Gingerbread Track at Lake St Clair (GIT) and Airstrip West (AIW) at Lake Pedder (Table 4) (M. Driessen unpublished data). To make the results from these studies directly applicable to current fire management practices in buttongrass moorlands, the design was based on two of the primary variables in the fuel and fire behaviour models currently used by PWS and described by Marsden-Smedley *et al.* (1999) (see Chapter 1). The first variable is vegetation age (i.e. years since fire). Although the predictive models do not include other fire regime parameters such as frequency and season, they are used in the prescription guidelines. As such, they are considered herein to the extent that reliable data were available for inferences. The second variable is site productivity as it pertains to fuel accumulation rates, loads, and spatial continuity. Sites are classified in the models as either low or moderate productivity based on geological substrate. Investigating productivity as it relates to intrinsic edaphic differences between sites appears to be a unique approach for an avian fire ecology study as neither the literature review by Woinarski (1999a, 1999b) nor that conducted for this thesis identified any such previous research (see References). While not an explicit model variable, the blanket and eastern moor community groups were also considered since they are closely associated with site productivity within the study area (i.e. low and moderate, respectively), and the latter group presents major issues for fire management (e.g. high fire spread rates and flame heights) (Marsden-Smedley and Catchpole 1995b; Marsden-Smedley *et al.* 1999). Although productivity is obviously a continuum and can vary over both time and space, and time since fire is likewise only one

factor in an extremely complex system, these variables have proven to be effective in modelling and predicting fuel loads and fire behaviour in buttongrass moorlands. Therefore, these two variables are used as part of the underlying paradigm of this study.

Site descriptions and fire regimes

Study locations

The island state of Tasmania (excluding associated islands) is located between latitudes 40°6'-43°6'S and longitudes 144°6'-148°4' E, approximately 200 km south of the Australian mainland. Research was conducted in the Tasmanian Wilderness World Heritage Area, associated National Parks and inholdings, and adjacent lands (e.g. State Forest and Hydro Electric Commission lands). The TWWHA was established in 1982 and encompasses 1.38 million hectares in western Tasmania (~ 20% of the State) (Smith and Banks 1993). The study comprises two locations, hereafter referred to as Lake Pedder (42°83'-43°05'S, 146°19'-146°38'E), primarily in Southwest National Park, and Lake St Clair (42°12'-42°21'S, 146°10'-146°22'E) primarily in Cradle Mountain-Lake St Clair and Franklin-Gordon Wild Rivers National Parks (Figure 1). These locations were chosen since sites near Lake Pedder are considered to be representative of typical blanket moorlands found throughout the low productivity quartzitic geologies of the Southwest, and sites near Lake St Clair are representative of typical eastern moorlands found in the moderate productivity doleritic geologies of the Central Plateau (Greenslade and Driessen 1999; Marsden-Smedley 2003). Furthermore, these regions are of high conservation concern (PWS 1999), are relatively accessible, and have been the focus of past and ongoing fire management activities and multidisciplinary fire ecology research (e.g. Marsden-Smedley 1993; PWS 1996; Driessen 1999; Driessen and Greenslade 2004; Barnes and Balmer 2000; Balmer and Barnes 2000; Bridle *et al.* 2003).

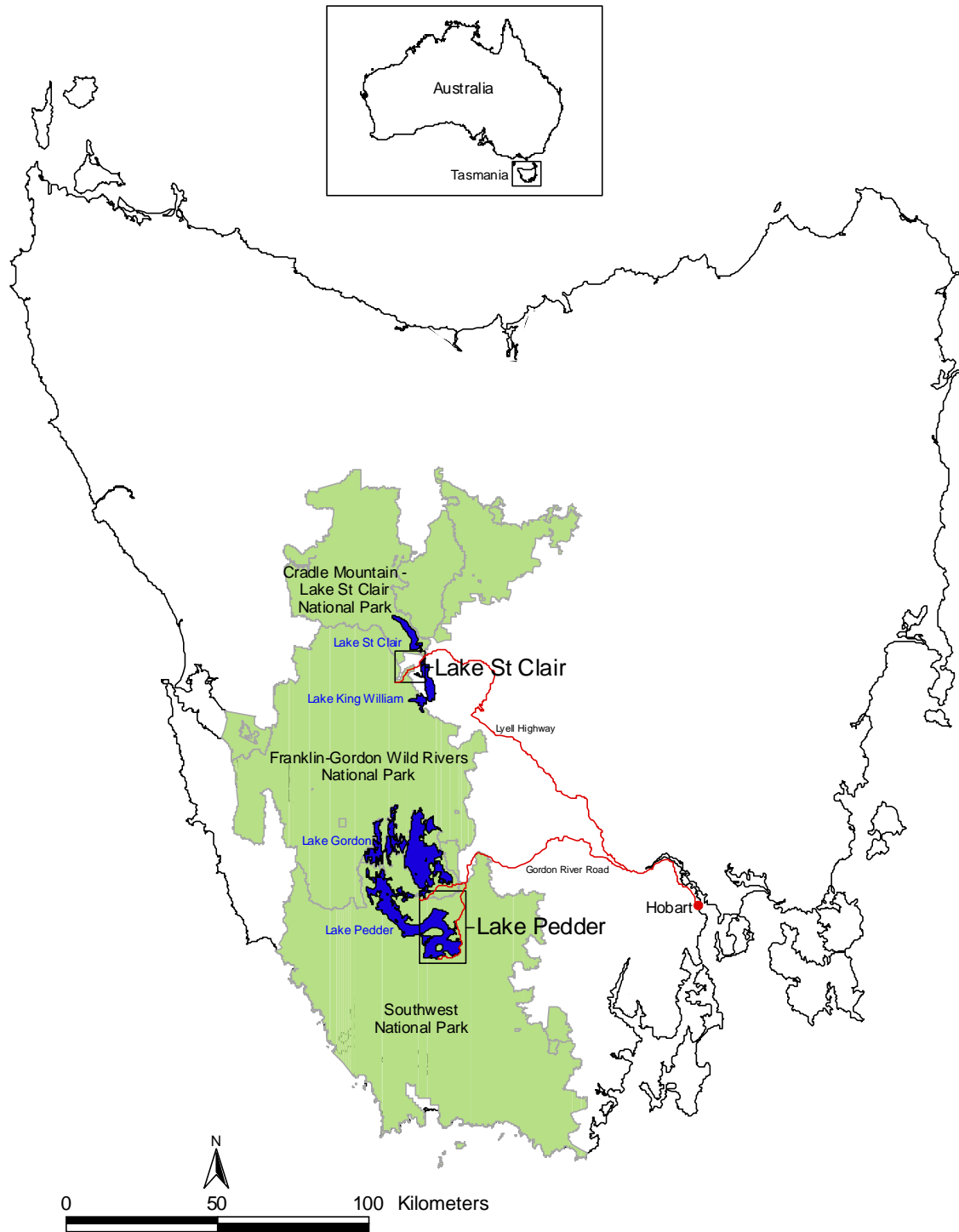


Fig. 1. Lake Pedder and Lake St Clair study locations, Tasmanian Wilderness World Heritage Area , Tasmania, Australia (AGD 66; LIST 2003). Note: King, Flinders, Macquarie, and associated islands not shown.

Geology and soils

Both study locations consist of vast tracts of wilderness with highly variable topography, ranging from broad plains (~ 300-700 m asl) to rugged, glacially-formed mountains (≤ 1450 m asl). They consist of a diverse range of structures, rock types, and soils reflecting a complex geological history (Hannan *et al.* 1993). The soils of the buttongrass moorland plains in this study generally consist of structureless, and to a lesser extent fibrous, autochthonous peats (i.e. organosols) (Jarman *et al.* 1988a). Mean peat depths are 39 ± 3 cm (range 21-67 cm) at Lake Pedder and 45 ± 3 cm (range 27-92 cm) at Lake St Clair (M. Driessen unpublished data). These peatlands are primarily underlain by Precambrian quartzite at Lake Pedder and Quaternary fluvio-glacial deposits and Jurassic dolerite at Lake St Clair; sites within these locations have been classified as low and moderate productivity in previous studies, respectively (Bowman *et al.* 1986; Pemberton 1986; Marsden-Smedley 1998a; Balmer and Barnes 2000).



Fig. 2. Typical soil profile of peat underlain by quartzite in blanket moorland at Lake Pedder.

Data on soil properties were collected as part of the collaborative research project conducted by the BCB (M. Driessen unpublished data). Four A horizon peat soil samples (70 cm diameter x 11 cm deep) were taken 20 m apart from a chronosequence of sites at Lake Pedder (3-65 years post-fire, $n = 19$) and Lake St Clair (1-31 years post-fire, $n = 25$); inclusive but not limited to all sites used in this study, except AIW and GIT (Table 4). Site samples were homogenised into one sample and analysed for the following soil properties using standard chemical testing: total Kjeldahl nitrogen, loss on ignition (LOI), pH, conductivity (Cond.), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), manganese (Mn), zinc (Zn), copper (Cu), boron (B), and soil moisture (SM) (Table 1).

Table 1. Summary of peat soil properties across fire ages for sites at Lake Pedder (3-65 years post-fire, $n = 19$) and Lake St Clair (1-31 years post-fire, $n = 25$), Tasmania (M. Driessen unpublished data).

Soil variables	Lake Pedder			Lake St Clair		
	Mean	SE	Range	Mean	SE	Range
Total Kjeldahl N (%)	0.73	± 0.07	0.33 - 1.41	1.25	± 0.06	0.42 - 1.73
LOI (%)	48	± 5	8 - 93	63	± 4	21 - 93
pH	3.8	± 0.0	3.5 - 4.0	4.1	± 0.0	3.8 - 4.7
Cond. (us/cm)	255	± 14	160 - 340	324	± 26	140 - 810
P (ppm)	8	± 1	4 - 14	12	± 1	6 - 32
K (ppm)	152	± 14	54 - 270	209	± 28	74 - 780
Ca (ppm)	357	± 36	170 - 630	821	± 101	300 - 2640
Mg (ppm)	532	± 66	180 - 1040	444	± 41	120 - 930
Mn (ppm)	6.7	± 0.2	5.0 - 8.1	55.5	± 24	8.3 - 520.0
Zn (ppm)	1.1	± 0.1	0.6 - 2.0	6.2	± 0.8	1.8 - 20.0
Cu (ppm)	0.4	± 0.0	0.2 - 0.6	5.7	± 0.9	1.4 - 15.0
B (ppm)	1.8	± 0.1	1.4 - 3.2	2.9	± 0.2	2.1 - 7.0
SM (%)	76.1	± 1.4	59.2 - 84.4	79.7	± 1.7	58.6 - 92.7

Principal Components Analysis (PCA) was used to identify overall differences in soil properties between Lake Pedder and Lake St Clair using Primer 5.2.2 (Primer-E Ltd. 2001). Data were normalised and log₁₀-transformed, as appropriate (Table 2) (Clarke and Warwick 1994; Clarke and Gorley 2001). The first two Principal Components (PC1 and PC 2) had eigenvalues > 1 (7.40 and 2.69, respectively) and accounted for 77.6% of the variation; thus, the two-dimensional PCA plot (Figure 3) provided a parsimonious summary and showed a clear difference in overall soil properties between Lake Pedder and Lake St Clair (Clarke and Gorley 2001; Quinn and Keough 2002).

A dissimilarity matrix was calculated using normalised Euclidean distances and a one way analysis of similarities routine (ANOSIM, with 999 permutations) was used to test for significant differences in soil properties between locations as depicted in the PCA plot (Clarke and Warwick 1994). The global ANOSIM test showed there were significant differences in overall soil properties between locations ($R = 0.498$, $P < 0.01$) and the moderate R value indicated there were clear, although overlapping differences (Clarke and Gorley 2001).

Overall, these results demonstrated there was a significant difference in soil properties between Lake Pedder and Lake St Clair. The sites at Lake Pedder generally have lower levels of soil nutrients and hence lower fertility and productivity than those at Lake St Clair and thus confirm their *a priori* classification as low and moderate productivity soils, respectively. Results from Lake Pedder are consistent with previous research by Bowman *et al.* (1986) that likewise concluded the soils in that region are extremely infertile.

Table 2. Principal Components Analysis eigenvectors of peat soil property values (log₁₀-transformed* and normalised from Table 1) for sites at Lake Pedder and Lake St Clair, Tasmania.

Soil variables	PC1	PC2
Total Kjeldahl N (%)	-0.313	0.028
LOI (%)	-0.283	0.250
pH*	-0.208	-0.391
Cond. (us/cm)*	-0.313	0.204
P (ppm)*	-0.296	0.205
K (ppm)*	-0.292	0.280
Ca (ppm)*	-0.312	-0.084
Mg (ppm)*	-0.206	0.398
Mn (ppm)*	-0.296	-0.241
Zn (ppm)*	-0.315	-0.241
Cu (ppm)*	-0.276	-0.325
B (ppm)*	-0.181	-0.437
SM (%)	-0.269	0.208

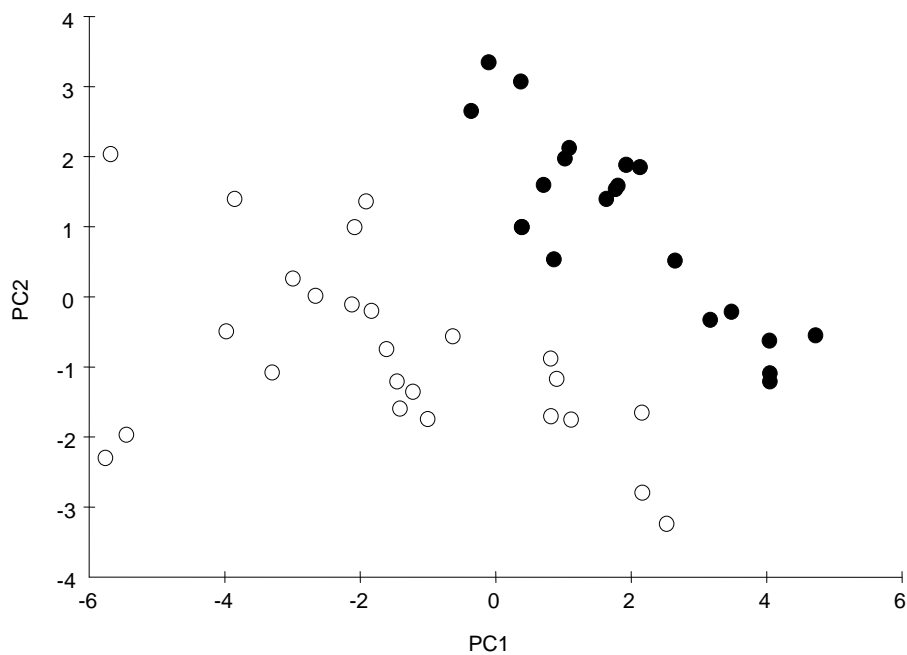


Fig. 3. Two-dimensional PCA ordination of the 13 peat soil property variables (Table 2) for sites at Lake Pedder (○ ; $n = 19$) and Lake St Clair (● ; $n = 25$), Tasmania (77.6% of variance explained).

Climate

Tasmania is characterised as having a temperate maritime climate dominated by the westerly ‘Roaring Forties’ wind system (ABS 2000). The Lake Pedder and Lake St Clair regions have cool to mild perhumid climates and are renowned for having extremely variable short-term weather conditions (Gentilli 1972). Table 3 includes seasonal and annual averages for climatic conditions obtained from the nearest and most representative weather stations for the study regions (Bridle *et al.* 2003).

Table 3. Average seasonal (based on bird survey periods) and annual climatic conditions for Lake Pedder (Scotts Peak Dam Station # 97083, 1992-2004) and Lake St Clair (Lake St Clair National Park Station # 096071, 1989-2004) study locations, Tasmania (BOM 2004).

Conditions	Lake Pedder				Lake St Clair			
	Feb.	Jun.	Oct.	Annual	Feb.	Jun.	Oct.	Annual
Temperature- daily max. (°C)	21.0	9.7	13.8	14.7	19.3	7.7	12.5	13.1
Temperature- daily min. (°C)	9.6	4.4	4.8	6.2	5.9	0.3	1.9	2.7
Relative humidity- 9am (%)	82.0	90.5	84.0	85.2	84.0	88.0	82.0	84.0
Precipitation (mm)	62.0	250.6	170.4	1819.9	85.6	193.2	180.5	1906.0
Rain days	14.4	25.5	24.8	256.7	12.4	22.9	22.4	235.6
Wind speed- 9am (km h ⁻¹)	6.7*	7.0*	9.6*	8.1*	5.0	5.0	6.4	5.5

* Mean wind speed from Strathgordon Village Station (# 097053, 1968-2004) located west of the study region since data not available for Scotts Peak Dam.

Climates in both regions are broadly comparable. Temperatures are normally cool to mild throughout the year, and range from an average daily maximum of approximately 19-21°C during summer to a minimum of 0-4°C during winter. Consistent with its higher altitude, minimum temperatures at Lake St Clair are typically a few degrees lower throughout the year than at Lake Pedder. Precipitation is highest in spring and winter and lowest in summer, though significant rain occurs throughout the year. Protracted periods without rain (i.e. ~ 2 weeks) are rare (Marsden-Smedley 1993), and extreme fire conditions generally occur for only up to a few days each summer (Jackson 1999a). A precipitation gradient decreasing from west to east exists in both study regions due to the predominantly westerly air flow and mountainous terrain (Nunez *et al.* 1996). Accordingly, mean precipitation figures for Lake St Clair are likely higher than indicated above since the data are from a station located in the eastern portion of the study area. Precipitation is sufficient to maintain rainforest communities in both locations (Balmer and Barnes 2000; TVMP 2004) and results in waterlogged conditions during most of the year in the poorly-drained peat flats in this study. Prevailing winds are from the northwest to southwest, but can be highly variable in speed, duration, and direction. Wind speeds are typically light (< 19 km h⁻¹) with occasional stronger gusts, but moderate to gale force winds (20-87 km h⁻¹) are not uncommon, particularly in winter and early spring (BOM 2004).

Site selection

The primary objective for the site selection process was to identify numerous replicate buttongrass moorland sites within each location that spanned a wide range of fire ages (i.e. chronosequences), encompassing the inter-fire interval range of 5 - \geq 30 years proposed for ecosystem-management burning in the TWWHA (Marsden-Smedley and Kirkpatrick 2000; PWS 2004; Marsden-Smedley 2009). Additional objectives included ensuring that sites had comparable abiotic and biotic characteristics, and that they were well interspersed within each study location.

In order to reduce the potential confounding effects of abiotic and biotic variables other than the primary treatment (i.e. fire age) both within and between sites, the following criteria were utilised during the site selection process:

- *Accessibility*

Due to a range of logistical and safety considerations, sites were initially limited to those that were either next to or within a reasonable walking distance of the relatively limited road networks found within the study area.

- *Site size*

Based on the estimated home ranges reported in the literature, five hectares was considered to be the minimum moorland patch size that could reasonably support at least one breeding pair for each of the three resident bird species (see Chapter 1). To ensure that each site could be surveyed in its entirety within a six hour survey period and to maximise the number of sites that could be surveyed during the course of the study the largest desirable site size was considered to be approximately 100 ha.

- *Altitude*

Altitude is known to affect a variety of abiotic factors that can, in turn, influence vegetation. Sites were limited to altitudinal ranges that are not known to contain any significant vegetation gradients (Jarman *et al.* 1988a; J. Marsden-Smedley pers. comm. 2003).

- *Topography*

Sites were limited to those that were flat to gently rolling (slope $\sim \leq 5\%$) in order to minimise the variability of site aspect and hydrology and their associated effects on the behaviour of past fire events, vegetation structure and composition, and bird habitat utilisation.

- *Geology*

Although the Lake Pedder and Lake St Clair study locations are characterised by quartzitic and doleritic geologies, respectively, localised variability in geology and associated productivity do occur (J. Marsden-Smedley pers. comm. 2003).

Accordingly, road-cuts, boulders, outcrops, and apparent fuel-loading (based on

estimated fire ages; see below) were examined to ensure that the sites under consideration were consistent with the type localities.

- *Climate and geography*

Sites were selected so that they were as geographically close to each other as possible to minimise climatic variability (e.g. along west-east precipitation gradients) and other spatial confounding factors between sites.

- *Fire history*

In order to partially control for variation in fire frequencies and inter-fire intervals, sites that had been burnt within ten years prior to the most recent known fire event, or were otherwise known to have been subjected to more frequent fires than the other available sites, were excluded from consideration, since ten years is considered to be an adequate inter-fire interval for both low and moderate productivity moorlands to vegetatively recover after fire (Gellie 1980; J. Marsden-Smedley pers. com. 2003).

- *Qualitative considerations*

Despite attempts to limit the variability of candidate sites by using the above criteria, it was necessary to eliminate some due to qualitative considerations, such as apparent differences in vegetation structure, composition, and hydrology.

Fire histories

Fire histories for the study regions have been constructed by Marsden-Smedley (1998a; 1998b) and Johnson and Marsden-Smedley (2002) based on published papers, fire history records, aerial photographs, and field data. Known fires from the 1930s to the present have been digitised as polygons at a 1:100,000-scale and are available in the Parks and Wildlife Service Fire History GIS layers (PWS unpublished data; Marsden-Smedley 1998a). Primary data fields include: fire name, date, ignition source, and estimated extent. Although this is the most comprehensive source for fire history data currently available, the quantity, quality, and scale of data are variable and no estimates of positional or attribute accuracy have been calculated. Furthermore, since the fire history layers are not definitive and do not include known fires prior to the 1930s, both the frequency and extent of fires within the study area have probably been underestimated (Johnson and Marsden-Smedley 2002). Since either no, or questionable, fire history data were available for many of the approximately 65 candidate sites that met the initial selection criteria, it was necessary to obtain data from additional sources to verify or determine fire ages and other regime parameters. Additional sources included aerial photographs, published and unpublished data, personal communication with local fire managers, and extensive field data.

Fire seasons for this study were determined from reported dates or inferred from other data and defined according to current fire management guidelines in Tasmania as follows: spring (August - mid-October), summer (mid-October - March), and autumn (April - June) (PWS 1996; Marsden-Smedley *et al.* 1999). Data on season and intensity of fires (kW m^{-1}) were limited overall and unavailable for some of the older sites. Data on ignition were also limited and typically only specified sources (e.g. planned or arson) and not actual methods and patterns of ignition (e.g. hand or aerial; line or point, respectively). In general, most planned burns over the past decade were conducted during the autumn and spring seasons since they typically have conditions most suitable for planned burning, resulting in fires of low to moderate intensity (Marsden-Smedley *et al.* 1999; T. Norris pers. comm. 2005). In contrast, older fire events that occurred during summer months were likely during conditions typically associated with more intense and extensive wildfires (Marsden-Smedley 1993).

Fire ageing

The age of buttongrass moorland sites is considered to be the time since the last major fire event consumed most of the above ground biomass (Jarman *et al.* 1988a). After a fire event, there is typically a pulse of vegetative and seed regeneration with the result that the aerial parts of plants are even-aged within most sites less than approximately 60 years of age (Jarman *et al.* 1988b; Marsden-Smedley 1990; Marsden-Smedley *et al.* 1999). Therefore, the age of the majority of sites can be estimated by counting the annual growth nodes (i.e. swelling at branch junctions) of *Banksia marginata* (Brown and Podger 1982) or the annual growth rings of a number of shrub and tree species, notably *Leptospermum* spp. which typically produce clear and countable rings (Jarman *et al.* 1988b) (Figure 4). The approximate fire age of a site, and hence the year of the last major fire event, is usually considered to be the modal count of samples plus one year (Marsden-Smedley *et al.* 1999). Wills (2002) assessed the reliability of the *Banksia* node count method in relation to sites of known fire ages in heathlands and demonstrated that it is an accurate (± 1 year) method to estimate site ages under most conditions.

The age of each site was either determined or ground-truthed by following the ageing method outlined by Marsden-Smedley *et al.* (1999). Preference was given to node counts since they are easily conducted and non-destructive. Since *Banksia marginata* does not occur within eastern moorlands at Lake St Clair, ring counts was the only available method. Ring count samples were obtained by cutting cross-sections of stems just above ground level with a hand saw. Due to the destructive nature of obtaining these samples and in consideration of the fact that the study sites were located within the TWWHA, the number of ring count samples that were collected for this study was necessarily limited ($n = 130$). At least six node and/or ring counts were obtained from each site, as recommended, but for sites

at Lake Pedder larger numbers of node counts were conducted ($n > 230$). Where possible, samples were taken throughout the site to ensure node and ring counts were consistent, particularly in areas where changes in vegetation patterns indicated possible fire boundaries. Ring count samples were dried, polished with fine-grade sandpaper, and annual growth rings were independently counted (by T. Chaudhry and M. Driessen) using a dissecting microscope. The approximate fire age of each site was estimated by comparing counts and calculating the mode plus one for these samples in conjunction with other available fire history data. The range of site fire ages was examined, and sites were grouped into logical fire age classes based on previous research that indicated significant differences in structure, floristics, and fuel loads between certain fire ages (Bryant 1991; Marsden-Smedley 1998a; Marsden-Smedley *et al.* 1999; Driessen 1999). Regardless of estimated age, most sites contained microhabitats, such as copses and riparian zones (see Chapter 3), which are less prone to burning than the surrounding moorland and thus may be older than indicated by the site fire age (Marsden-Smedley 1990).



Fig. 4. *Banksia marginata* stem showing annual growth nodes at Lake Pedder (left) and an unsanded cross-section of a *Leptospermum lanigerum* stem showing annual growth rings at Lake St Clair (right).

Site mapping and data compilation

Sites were delineated using a number of methods and it was assumed that during the most recent fire event the majority of the aboveground biomass within the buttongrass moorland was scorched, partially burnt, or fully consumed within each site as defined. Younger sites ($\sim \leq 10$ years since fire) could be delineated with a high level of accuracy due to the availability of fire maps, the presence of fire-killed or scorched woody vegetation, and regrowth at the sites (Figure 5), as well as personal communication with fire managers



Fig. 5. Boundary between 3 year old McPartlan Escape West site (MEW, foreground) and 31 year old Wedge Inlet site (WEI, background) in blanket moorland, Lake Pedder, Tasmania. Note the reduced cover of graminoids and especially shrubs in MEW.

involved with the site-specific fire operations. Older sites were delineated by referring to coarse-scale fire maps when available, by determining the area over which node and/or ring counts were consistent within a contiguous patch of moorland, and by identifying known or likely fire boundaries during the most recent and past fire events. Such secure fire boundaries included both natural and man-made firebreaks such as woodland edges, roads, and watercourses (Marsden-Smedley *et al.* 1999). For the latter, the watercourse itself was considered to form the actual boundary, so that only the riparian vegetation on the inside of the watercourse was included in the site. Some site boundaries were also defined by significant changes in topography, geology, or maximum site size *as per* the site selection criteria. Therefore, the calculated metrics (e.g. area and perimeter), particularly for older sites, did not necessarily coincide with the actual extent of the last fire event.

The final sites were mapped as polygons in ArcView GIS 3.3 (ESRI Inc. 2002) by digitising their boundaries from orthorectified aerial photographs (see Chapter 3). Site names were derived from major geographical features from the 1:25,000 topographical maps (TASMAP 2003). Site locations were denoted by the Universal Transverse Mercator (UTM) coordinates of the first bird survey point at each site as well as presented on 1:100,000 digital topographic maps (TASMAP 2003). Mean altitudes (m asl) of sites were calculated to the nearest 5 m from the 1:25,000 digital contour layer (LIST 2003). Standard geoprocessing techniques were used to calculate site areas (ha) and perimeters (m), the estimated area (ha) of sites burnt, and the estimated total extent (ha) of fire events based on the study site

polygons and the fire polygons included in the Parks and Wildlife Service Fire History GIS layers (PWS unpublished data; Marsden-Smedley 1998a). Fire names, dates, and ignition sources were obtained from the PWS layers as well, except for sites that were solely aged in this study from node and/or ring counts or from other sources. In such cases, the fire events were named after the sites, ages were approximated to the nearest year, the estimated area of the site burnt was assumed to be the same as the total area of the site as defined, and the ignition sources and the total extent of these fire events were reported as unknown.

Study sites

A total of 26 sites were chosen based on the selection criteria and fire ageing data (Table 4, Figures 6-8). The sites included 12 at Lake Pedder ranging in age from 3-54 years post-fire, and 14 at Lake St Clair ranging from 1-44 years post-fire. These ages span the regrowth (< 15 years), mature (15- 35 years), and old-growth (> 35 years) fire age classes identified by Marsden-Smedley (1998a) for blanket moorlands. Classes at a finer resolution were desirable for some of the subsequent analyses so the following eight classes were utilised with the years since fire shown in parenthesis: 1 (1), 2 (2-3), 3 (5-8), 4 (12), 5 (14-16), 6 (22-27), 7 (30-33), 8 (44-54). Detailed fire history data (from 1933-2005) for the study sites are presented in Table 5. Replicate sites were identified for all fire age classes, except class 4 at Lake Pedder, and classes 1 and 8 at Lake St Clair, due to a lack of availability. Replicates were considered to be sites within the same fire age class that were either burnt during different fire events (e.g. NPW and KWE) or were burnt on the same date but were spatially separated at an appropriate scale in relation to the expected small home ranges and/or limited dispersal capabilities of the resident species (e.g. > 2 km between MEW and MEE) (see Chapter 1). Within a SFT context, the latter cases may be considered pseudoreplicates in the strict sense of the term (Hurlbert 1984). However, fire events (i.e. treatments) in general are inherently variable over space and time (Clark *et al.* 2002). Fire behaviour research and personal observations confirm such spatiotemporal variability in buttongrass moorlands (Marsden-Smedley 1993). In other words, both fire behaviour and site conditions are often variable at relatively fine scales (i.e. metres or minutes) such that the scale of the sites (i.e. hectares) and the separation between sites (i.e. kilometres) warranted treating them as independent replicates.

Although data for all known fire events are presented in Table 5, fire frequencies were not calculated due to a lack of adequate data and the inability to verify the mapping of historical burns at the site scale. It was apparent that most sites have burnt at frequencies that are consistent with the maintenance of moorland communities of the Southwest (Jackson 1968), while it is assumed that this is likely the case for other sites for which no historical data were available (e.g. SCS, SCN, TRR). However, other factors such as frost and hydrology may

also influence their perpetuation, particularly regarding eastern moorlands at Lake St Clair (Jarman *et al.* 1988a). While it may appear as though some sites have relatively short inter-fire intervals (e.g. KWW, Table 5), the PWS fire maps indicated that many of these events only burnt small portions of such sites. Years of some of the major fire summer events that likely affected some of the study sites, including 1933/34, 1960/61, 1980/81 and 1981/82, occurred during dry years or longer-term drought conditions (Marsden-Smedley 1998a). Although fires prior to 1930 have not been mapped, it is known that extensive fires also took place before the 1930s, particularly in 1851 and 1897/98, as well as extensive aboriginal burning prior to European colonisation, but no detailed information of an acceptable reliability and scale is available (Marsden-Smedley 1998a; see Chapter 1).

Overall, the availability of fire history data for the TWWHA and extensive ground-truthing within the moorlands enabled a relatively sound characterisation of fire regimes for the study area for use in this and subsequent studies. It was therefore possible to develop a replicated SFT study spanning a broad range of moorland fire ages within two locations that primarily differ in respect to soil productivity, moorland type, and altitude. This is in contrast to most previous studies on avian fire ecology in Australia that have included only limited chronosequences with little or no replication (Woinarski 1999a; 1999b). However, reliable and consistent data are still lacking (see Chapter 7) and it is apparent that fire regimes within the study area are heterogeneous at multiple spatiotemporal scales, reflecting a long history of anthropogenic burning and complex interactions between fire, soils, topography, climate, and vegetation. Details on vegetation composition, structure, and spatial characteristics of the study sites are provided in Chapter 3.

Table 4. Study sites in Tasmania by location and fire age class. Fire ages (as of 2004) are based on records in Table 5. Easting/Northing are UTM coordinates (AGD 66) and are for the first survey point at each site. Altitudes are mean asl within study sites to the nearest 5 m. Area and perimeter estimates based on habitat mapping described in Chapter 3.

Location (Productivity)	Fire age class (Range)	Site name	Site ID	Fire age (Years since fire)	Fire season	Easting (UTM)	Northing (UTM)	Altitude (m asl)	Area (ha)	Perimeter (m)
Lake Pedder (Low)	2 (2-3)	McPartlan Escape East	MEE	3	SPR	438068	5256238	320	53.33	5973
		McPartlan Escape West	MEW	3	SPR	435549	5255753	315	48.94	3927
	4 (12)	McPartlan Research	MCR	12	SPR-SUM-AUT	435450	5255002	330	21.22	2748
	6 (22-27)	Sandfly Creek	SAC	22	SPR	449101	5250946	330	31.48	2304
		Condominium Creek Southeast	CCS	25	SUM	448267	5241634	345	33.09	3105
		Condominium Creek Northwest	CCW	27	SPR or AUT	447418	5243913	325	23.92	2521
	7 (30-33)	Condominium Creek North	CCN	31	UNK	447531	5244069	355	15.18	1871
		Wedge Inlet	WEI	31	SUM	435682	5255915	310	38.76	3460
		Gelignite Creek	GEC	33	SUM	449094	5252606	370	20.20	2378
	8 (44-54)	Airstrip East	AIE	54	SUM	445959	5234734	310	13.62	1958
		Airstrip West	AIW	54	SUM	445969	5234506	305	11.75	1698
		Edgar	EDG	54	SUM	447661	5236414	350	13.12	1874
Lake St Clair (Moderate)	1 (1)	Harbacks Road	HAR	1	SPR	430686	5328297	745	32.65	3441
	2 (2-3)	Lake St Clair Road North	SCN	2	AUT	434542	5335872	730	7.61	1799
		King William Creek East	KWE	3	AUT	430088	5329254	750	22.48	3072
		Navarre Plains West	NPW	3	SUM	430915	5330267	735	16.98	2568
	3 (5-8)	Beehive Canal South	BCS	5	AUT	426660	5326465	790	83.45	6837
		King William Creek West	KWW	5	AUT	429066	5327557	775	45.39	6280
		Lake St Clair Road South	SCS	8	AUT	435623	5334736	730	21.67	4317
	5 (14-16)	Bedlam	BED	14	UNK	432022	5335959	745	19.79	3091
		Flood Creek	FLC	14	UNK	434059	5331608	745	47.68	5564
		Navarre Plains East	NPE	16	SUM	431625	5330349	730	70.58	6309
	7 (30-33)	Beehive Canal North	BCN	30	UNK	426907	5327849	775	23.73	3207
		Rufus Canal Road	RCR	30	UNK	427998	5331255	795	18.95	2699
		Travellers Rest	TRR	31	UNK	436154	5335360	730	39.04	3731
	8 (44-54)	Gingerbread Track	GIT	44	UNK	428533	5332458	890	6.89	1520

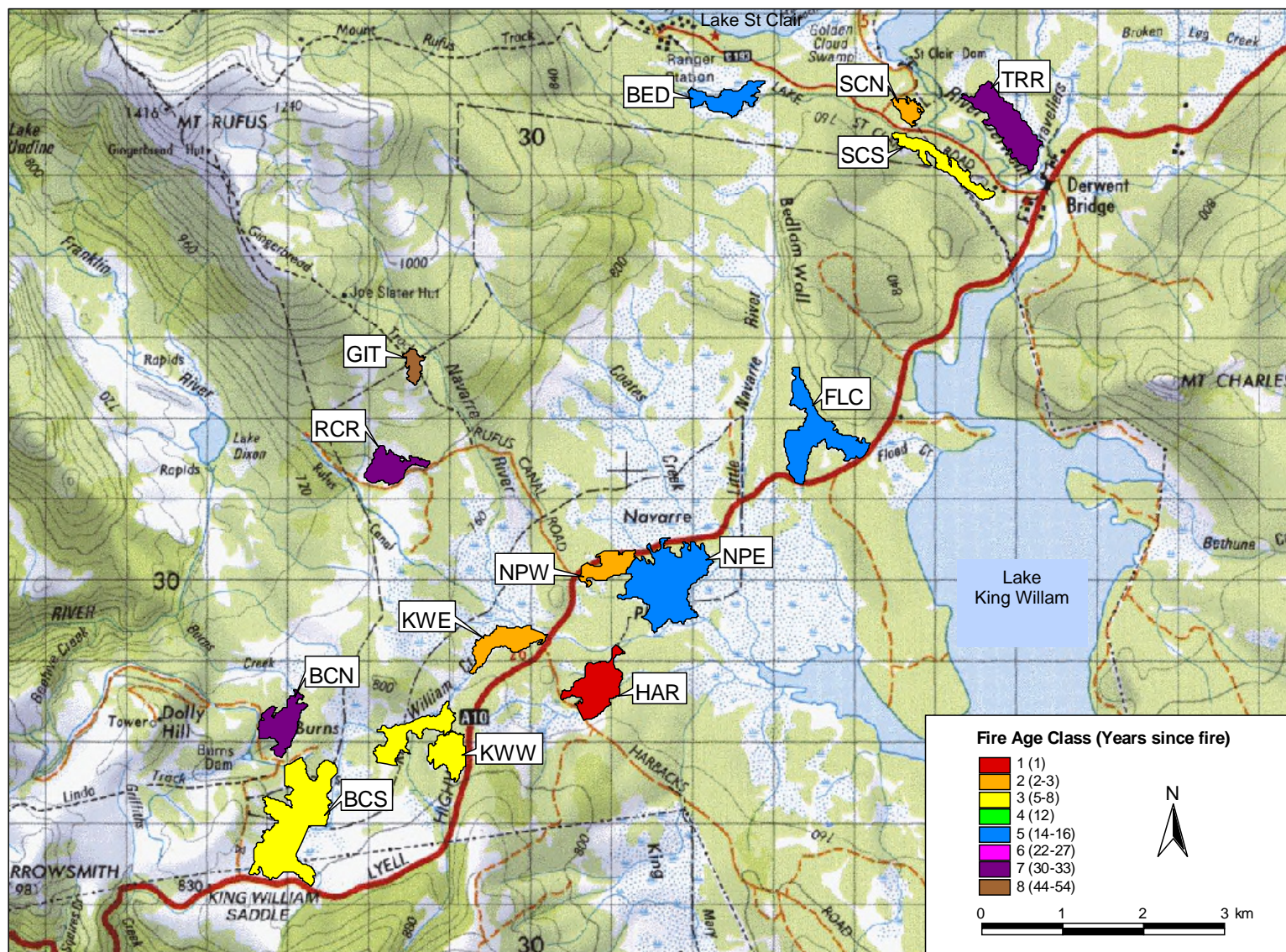


Fig. 6. Study sites by fire age class (as of 2004), Lake St Clair region, Tasmanian Wilderness World Heritage Area (1:100,000 Sheet #8113; AGD 66; TASMAT 2003). Note NPE and TRR were burnt by PWS in 2005 and were thus reclassified as fire age class 1 for the BACI study presented in Chapter 6.

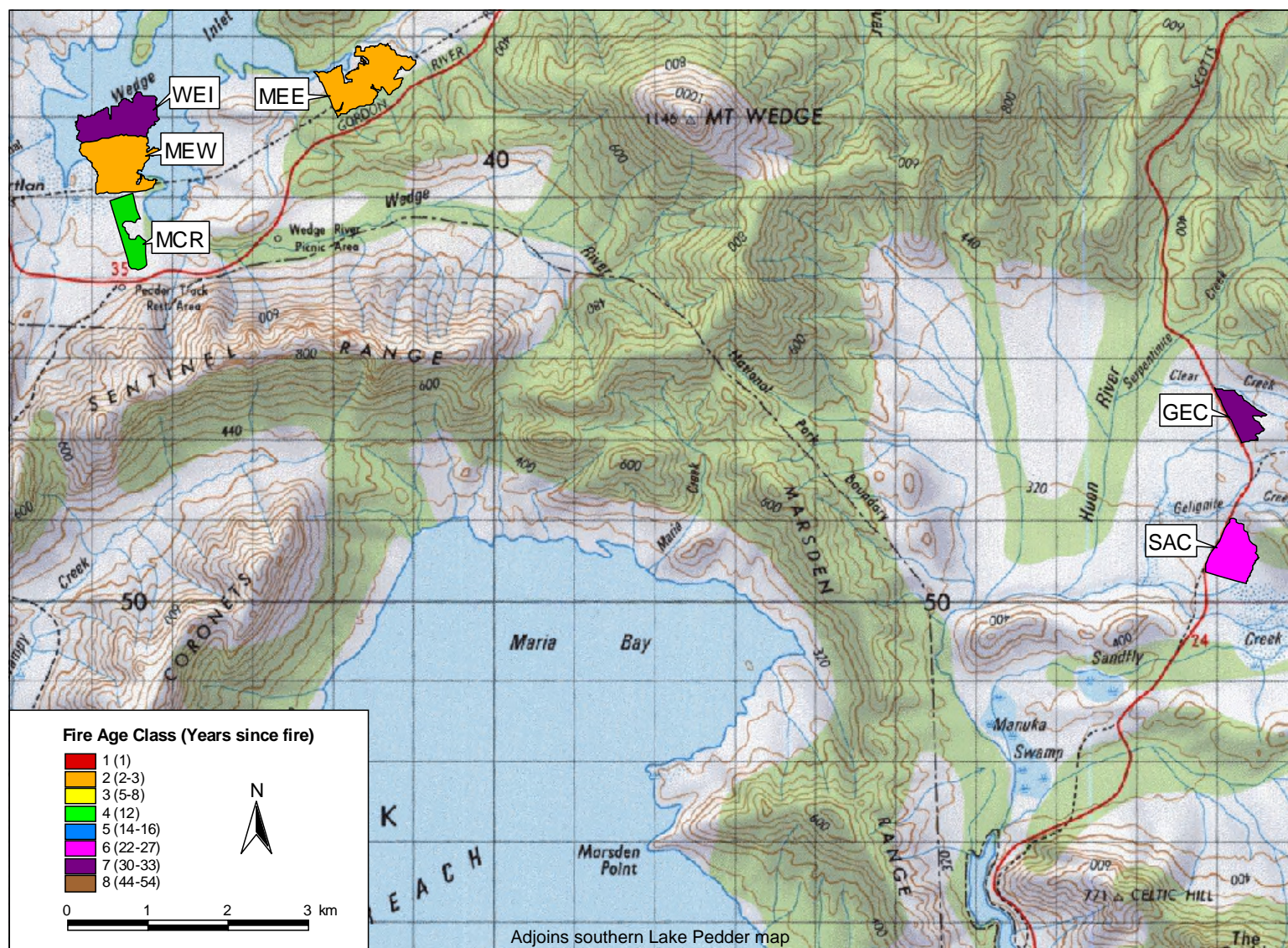


Fig. 7. Study sites by fire age class (as of 2004), northern Lake Pedder region, Tasmanian Wilderness World Heritage Area (1:100,000 Sheet #8112; AGD 66; TASMAT 2003).

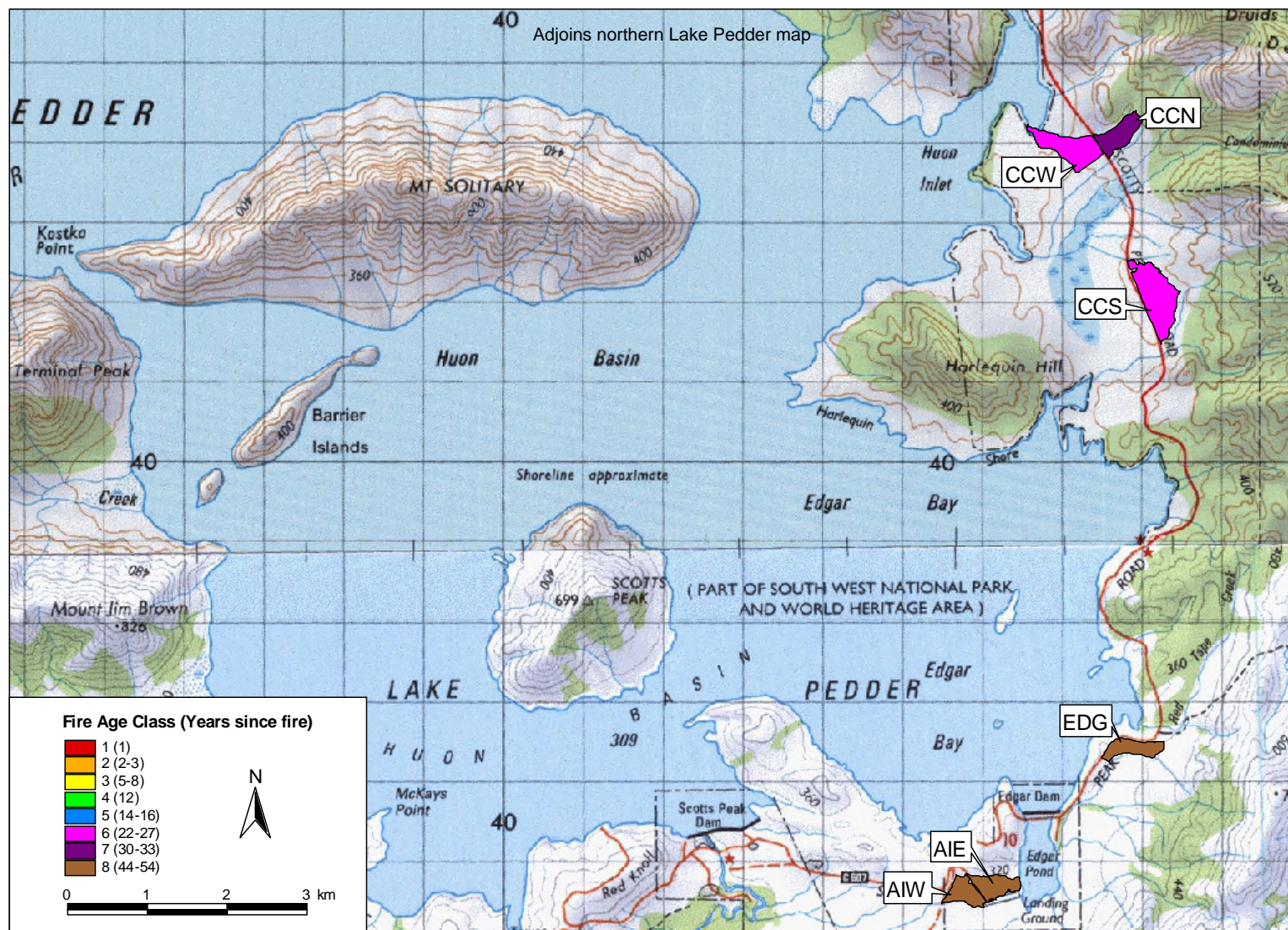


Fig. 8. Study sites by fire age class (as of 2004), southern Lake Pedder region, Tasmanian Wilderness World Heritage Area (1:100,000 Sheets #8112/8111; AGD 66; TASMAT 2003).

Table 5. Fire history records for study sites at Lake Pedder and Lake St Clair (1933-2005), Tasmania. Records are in chronological order and bold records are the most recent fire events used to define site ages for the SFT study (Chapters 4-5) and *italics* records are reburnt sites included in the BACI study (Chapter 6). Ignition source: PLA- Planned burn, ESC- Escaped, ARS- Arson, UNK- Unknown. Source: ^aThis study (based on ring/node counts and ground-truthing); ^bParks and Wildlife Service unpublished data; ^cMarsden-Smedley 1998a; ^dMarsden-Smedley 1993; ^eMarsden-Smedley 1990; ^fParks and Wildlife Service 1996; ^gDriessen and Greenslade 2004; ^hBalmer and Barnes 2000; ⁱGellie 1980; ^jForestry Tasmania unpublished data.

Lake Pedder						
Site ID	Fire name	Date (dd/mm/yr or est. yr)	Ignition source	Est. area (ha) of site burnt	Est. area (ha) of fire event	Source
MEE	Wedge Inlet	11/10/2001	ESC	53.34	666.54	a,b,c
	Airstrip Road research burns	27/10/1992-16/11/1992	PLA	4.90	4.90	b,c,d
	Lake Gordon (Starfish fire)	1972/73	UNK	52.71	30537.30	b,c
	1933/34 fire	1933/34	UNK	52.10	573187.70	b,c
MEW	Wedge Inlet	11/10/2001	ESC	48.94	666.54	a,b,c
	Lake Gordon (Starfish fire)	1972/73	UNK	48.94	30537.30	b,c
	1933/34 fire	1933/34	UNK	48.94	573187.70	b,c
MCR	McPartlan Pass research burns	28/10/1991-14/11/1992	PLA	21.22	21.22	a,d
	Lake Gordon (Starfish fire)	1972/73	UNK	21.22	30537.30	b,c
	1933/34 fire	1933/34	UNK	21.22	573187.70	b,c
SAC	Sandfly Creek	9/1982	PLA	31.48	222.12	a,b,c
	Upper Huon River	1/11/1971	PLA	31.48	2374.71	b,c
	Sandfly Creek	16/12/1970	PLA	18.89	259.25	b,c
	Lake Pedder	1950	UNK	31.48	27112.40	b,c
	1933/34 fire	1933/34	UNK	31.48	573187.70	b,c
CCS	Harlequin Hill	1978/79	UNK	33.06	673.40	a,b,c
	Lake Pedder	1950	UNK	33.06	27112.40	b,c
	1933/34 fire	1933/34	UNK	33.06	573187.70	b,c
CCW	Harlequin Hill	1978/79	UNK	1.75	673.40	b,c
	Condominium Creek	1977	PLA	23.92	135.16	a,b,c,i
	Lake Pedder	1950	UNK	23.92	27112.40	b,c
	1933/34 fire	1933/34	UNK	23.92	573187.70	b,c
CCN	Harlequin Hill	1978/79	UNK	0.92	673.40	b,c
	Condominium Creek	1977	PLA	2.00	135.16	b,c
	Celtic Hill	24/06/1975	PLA	0.24	140.09	b,c
	Condominium Creek North	1972/73	UNK	15.18	UNK	a,b,c
	Lake Pedder	1950	UNK	13.28	27112.40	b,c
	1933/34 fire	1933/34	UNK	13.12	573187.70	b,c
WEI	Lake Gordon (Starfish fire)	1972/73	UNK	38.77	30537.30	a,b,c
	1933/34 fire	1933/34	UNK	38.77	573187.70	b,c
GEC	Upper Huon River	1/11/1971	PLA	20.20	2374.71	a,e
	Lake Pedder	1950	UNK	20.20	27112.40	b,c
	1933/34 fire	1933/34	UNK	20.20	573187.70	b,c
AIE	Edgar Pond	1971/72	UNK	UNK	52.00	a,b,c
	Lake Pedder	1950	UNK	13.62	27112.40	a,b,c
	1933/34 fire	1933/34	UNK	13.62	573187.70	b,c
AIW	Edgar Pond	1971/72	UNK	1.12	52.00	b,c
	Lake Pedder	1950	UNK	11.75	27112.40	a,b,c
	1933/34 fire	1933/34	UNK	11.75	573187.70	b,c
EDG	Lake Pedder	1950	UNK	13.12	27112.40	a,b,c
	1933/34 fire	1933/34	UNK	13.12	573187.70	b,c

Table 5. cont.

Lake St Clair						
Site ID	Fire name	Date (dd/mm/yr or est. yr)	Ignition source	Est. area (ha) of site burnt	Est. area (ha) of fire event	Source
HAR	Harbacks Road	14/10/2003	PLA	32.65	115.25	a,b,c
	King William Plains	10/12/1989	ARS	4.71	94.86	b,c
	King William Plains	1979	UNK	5.12	2773.00	b,c
	1933/34 fire	1933/34	UNK	32.65	573187.70	b,c
SCN	Lake St Clair Road North	30/4/2002	PLA	7.61	7.61	b
KWE	King William Creek	4/5/2001	PLA	22.48	22.48	b,c
	King William Creek	1987	PLA	22.48	22.48	b,c
	King William Plains	1979	UNK	13.87	2773.00	b,c
	King William Creek	1977/78	UNK	0.09	11.30	b,c
	1933/34 fire	1933/34	UNK	22.48	573187.70	b,c
NPW	Lyell Hwy/Rufus Canal Road	5/2/2001	ARS	16.98	19.93	b,c
	Little Navarre River	12/1/1997	UNK	0.20	462.71	b,c
	Coates Creek	12/11/1988	UNK	16.98	152.82	b,c
	1933/34 fire	1933/34	UNK	16.98	573187.70	b,c
BCS	Beehive Canal	12/4/1999	PLA	83.45	313.00	b,c
	Beehive Canal	1987	PLA	83.45	231.42	b,c
	King William Plains	1979	UNK	UNK	2773.00	b,c
	1933/34 fire	1933/34	UNK	83.45	573187.70	b,c
KWW	King William Creek	17/4/1999	PLA	44.64	44.64	b,h
	Beehive Canal	4/1/1999	PLA	2.32	231.42	b,c
	Beehive Canal	1987	PLA	45.39	146.00	f
	King William Creek	9/3/1984	ARS	16.42	66.17	b,c,h
	King William Plains	1979	UNK	42.88	2773.00	b,c
	King William Creek	1977/78	UNK	11.46	176.40	b,c
	1933/34 fire	1933/34	UNK	45.39	573187.70	b,c
SCS	Lake St Clair Road South	26/9/1995- 4/5/1996	PLA	21.67	21.67	a,b
BED	Bedlam	1990	UNK	19.79	UNK	a
	1933/34 fire	1933/34	UNK	19.79	573187.70	b,c
FLC	Little Navarre River	1989/1990	UNK	47.68	UNK	a,b,j
	Coates Creek	1980	UNK	10.36	63.49	b,c
	1933/34 fire	1933/34	UNK	41.49	573187.70	b,c
NPE	<i>Navarre Plains</i>	<i>9/21/2005</i>	<i>PLA</i>	<i>66.15</i>	<i>611.79</i>	<i>a,b</i>
	Little Navarre River	12/1/1997	UNK	0.50	462.71	b,c
	Coates Creek	12/11/1988	UNK	70.58	152.82	a,b,c
	1933/34 fire	1933/34	UNK	70.58	573187.70	b,c
BCN	King William Saddle	1974	UNK	23.72	2773.00	a,b,c
	1933/34 fire	1933/34	UNK	23.72	573187.70	b,c
RCR	Rufus Canal Road	1974	UNK	18.95	UNK	a,g
	1933/34 fire	1933/34	UNK	18.13	573187.70	b,c
TRR	<i>Travellers Rest</i>	<i>4/21/2005</i>	<i>PLA</i>	<i>36.20</i>	<i>41.97</i>	<i>a,b</i>
	Travellers Rest	1973	UNK	39.04	UNK	a
GIT	Gingerbread Track	1960	UNK	6.89	UNK	a

Chapter 3

Avian habitat mapping in buttongrass moorlands and an accuracy assessment of the Tasmanian Wilderness World Heritage Area Vegetation Mapping

Introduction

The characteristic biotic and abiotic features of the environment in which a given species lives define its habitat, and provide the proximate or ultimate resources required for survival and reproduction. For birds in particular, it has long been recognised that both vegetation structure (e.g. MacArthur 1958) and floristics (e.g. Rotenberry 1985) can play critical roles in determining the composition and relative abundance of communities. This recognition has guided much of the research on avian habitat relationships to date (for reviews see Hildén 1965; Block and Brennan 1993; Jones 2001). The relative importance of vegetation structure and floristics on avian communities has been the focus of much debate; however, results may ultimately be dependent upon both the temporal and spatial scales at which a given system is being studied (Wiens *et al.* 1987; Mac Nally 1990; Collins 2001). Furthermore, it is important to recognise that habitat selection is a hierarchical process, such that birds select habitat at scales ranging from the geographic or home range (i.e. macrohabitat) to specific elements within their home range (i.e. microhabitat) (Hildén 1965; Johnson 1980; Block and Brennan 1993). To determine actual preference for a particular habitat, it is necessary to quantify both the relative usage (i.e. quantity utilised) and spatiotemporal availability (i.e. accessibility) of habitats at the relevant scales of interest (*sensu* Johnson 1980; see Chapter 5). Thus the objectives, design, and subsequent conclusions of a study are not only contingent upon what variables are measured, but also on the scales those measurements are made in relation to those of the underlying processes.

Relatively few studies to date, including those conducted on the Australian mainland, have focused on the habitat relationships of the resident bird species found in Tasmanian moorlands, namely the Southern Emu-wren, Striated Fieldwren, and Ground Parrot (see Chapter 1). These studies have primarily centered on defining the microhabitat of these species by identifying environmental features (e.g. vegetation floristics and structure) that were used to meet (at least proximally) their resource needs. These studies typically consisted of qualitative assessments of occupied habitat (e.g. Fletcher 1913a; Gosper and Baker 1997) or quantitative vegetation surveys (e.g. Bryant 1991; McFarland 1991a). The baseline information obtained from these studies is critical to understand the ecology of these species. However, considering their methods, study locations, and scale, this information

cannot be readily applied to studies such as the present one that investigates the influence of spatial heterogeneity and the underlying landscape-scale processes (e.g. fire regimes) on avian communities. Since conservation and fire management decisions are increasingly being made at the ecosystem level in the TWWHA (e.g. PWS 1999; Marsden-Smedley and Kirkpatrick 2000), it is imperative to provide information to land managers on avian habitat relationships that are location-specific and at a commensurate scale.

One way to bridge the gap between fine-scale, microhabitat and coarse-scale, macrohabitat characterisations is to utilise remote sensing and geographic information systems (GIS) to develop a habitat map. Although any map is essentially an imperfect model of reality (Foody 2002), maps are spatially continuous and thus may be used to provide a more comprehensive description of the environment than the sample-based vegetation surveys described above. These technologies also allow for vegetation mapping to be conducted, analysed, and displayed at varying spatial scales, and are increasingly relied upon in both bird conservation (e.g. Buchanan *et al.* 2005) and fire management (e.g. Bobbe *et al.* 2001). These methods are particularly powerful when used in conjunction with field-based avian surveys since they provide a means of quantifying the relationships between birds and the spatial heterogeneity of their habitats (i.e. composition, structure, spatial configuration) (McGarigal and McComb 1995; Li and Reynolds 1995; Fuller *et al.* 2005). Provided that both bird abundance and habitat usage data are collected, and mapping is at a fine enough scale and of an acceptable accuracy, it is possible to gain a more holistic understanding of avian habitat relationships and their underlying processes. These relationships can then potentially be extrapolated to even larger scales, which is a critical step towards understanding and managing natural systems (Bibby *et al.* 2000).

Despite the increasing use of remote sensing and GIS technologies reported in the wildlife literature, a review by Glenn and Ripple (2004) revealed that most papers failed to report important information regarding the digital maps that were used to assess wildlife habitat. For example, 23% of studies ($n = 44$) did not cite their map data sources; 55% did not specify the map scale, pixel size, or minimal mapping unit; 80% did not report map accuracy; and 27% utilised 'off-the-shelf' map products developed by other parties. Without adequate documentation it is difficult to interpret and assess the results of such studies, particularly as they pertain to determining habitat use and selection.

To avoid potential biases associated with the unquestioning use of an existing map product as the sole basis for classifying and quantifying vegetation within the study area, it was considered desirable to assess the accuracy of the existing Tasmanian Wilderness World Heritage Area Vegetation Mapping (hereafter WHAveg) (TVMP 2004) in relation to the

overall aims of this study. A brief overview of WHAveg is presented below, while a formal accuracy assessment is described herein.

WHAveg is a relatively coarse-scale, remote sensing, and polygon-based vegetation classification and mapping project covering the 1.38 million ha of the TWWHA (as compared to the less than 800 ha covered by this study) and was intended to provide detailed regional information on vegetation communities for natural resource management and research (ILS 2004; TVMP 2004). The WHAveg mapping methodology and classification system has changed over time and across different locations within the TWWHA. The following description is based on the metadata for WHAveg provided in the Tasmanian Spatial Data Directory (ILS 2004; TVMP 2004). A complete 1:25,000 colour run of aerial photographs was taken in 1988 on which most of the vegetation mapping has been based. Prior to 2000, a more complex synusia-based (i.e. associations of distinct strata; $n = 534$) classification scheme was used largely in the eastern TWWHA (including most of the study area). Aerial photographs were interpreted and the different vegetation communities were delineated with a grease pen onto transparencies overlain on each photograph. The resulting vegetation class polygons were ocularly transferred onto topographic base maps, scanned, imported into GenaMap GIS (Genaware Pty. Ltd. 1995), and labelled. This process included extensive field verification. After 2000, a simpler community-based classification scheme (i.e. vegetation types; $n = 55$) was developed for the more remote western TWWHA. The photographic interpretation was similar to the above method; however, vegetation communities were delineated with a fine marker onto transparencies. The resulting vegetation polygons were then scanned and orthorectified using proprietary photogrammetry PhotoFactory software (SonarData Pty. Ltd. 2005), including a Triangulated Interlaced Network Elevation Model, and then imported into GenaMap. These have subsequently been converted to shapefiles and stored in ArcView 9.0 (ESRI Inc. 2002). Only limited field verification has been conducted. Accordingly, attribute accuracy reported in the metadata is considered to be variable (and has not been quantified to date), with higher accuracy assumed to be in areas classified under the former scheme. Positional accuracy is qualitatively estimated at 50 m; however, the latter method is considered to be relatively more accurate.

To make these different classification systems compatible across the TWWHA, a set of new, mutually exclusive amalgamated classification codes was developed and is used as the base for the classification used herein (TVMP 2004). As with many maps, it should be noted that this system uses broad vegetation classes that represent communities that can be floristically and structurally variable even within a given vegetation patch (Jarman and Crowden 1978). For example, some copse communities were originally classified and delineated separately in

the synusia-based classification scheme, but were subsequently amalgamated into their surrounding polygons and associated classes. Considering such simplifications inherent in any coarse-scale map product, WHAveg and its derivatives should be interpreted with some caution. After an initial assessment of the WHAveg GIS layer, it was obvious that significant delineation, classification, and registration errors existed and that the scale of mapping and level of classification would not entirely meet the needs of this study. For example, polygons representing the various vegetation classes were often offset and simplified (i.e. smoothed) so that the area covered by a given polygon and the shape of its boundaries with adjacent polygons did not accurately reflect the boundaries on the orthorectified aerial photographs or those known to exist on the ground. In addition, features such as copses and riparian vegetation zones were not typically mapped since they were not considered part of their overall mapping objectives, were not readily identified from the aerial photos, or were below the minimal mapping unit (~ 0.5 ha within the study area). Accordingly, a formal accuracy assessment was conducted on WHAveg, which to the knowledge of the author is the first of its kind since neither the Tasmanian Vegetation Mapping and Monitoring Program (BCB) nor any end users have conducted a formal assessment to date. Knowing the potential limitations of the acquired layers *a priori*, it was also deemed necessary to delineate site boundaries and habitat features at a finer scale and greater resolution so that subsequent quantification of the habitat metrics would be more inclusive, accurate (> 85% attribute accuracy; Congalton and Green 1999), and appropriate for the aims of this study. Finally, it should be noted that this study was completed before the recent publication of the State-wide TASVEG Version 2.0 mapping scheme that was, in part, derived from the WHAveg mapping (TVMMP 2009). Consequently, the TASVEG communities and associated codes are not explicitly referenced herein.

The primary aims of this study were: 1) to develop a fine-scale and high-resolution classified avian habitat map for buttongrass moorlands and associated vegetation communities using remote sensing and GIS techniques; 2) to perform an accuracy assessment of WHAveg within the study area; 3) to quantify the composition, structure, and spatial characteristics of avian habitat for use in the analyses presented in Chapters 4-6; and 4) to provide a foundation for the potential future development of landscape-scale bird-habitat prediction models in buttongrass moorlands.

Methods

Image acquisition and processing

Aerial photographs were used as the basis for remote sensing in this project since they were readily available, provided an adequate spatial resolution, and were used for the original WHAveg classification. The most recent complete run of colour aerial photographs covering each study location was examined for site and habitat mapping (Table 1) (TASMAP 1988, 1993).

Table 1. Orthorectified aerial photographs, including maximum positional errors (m).

Location	Sites covered	Date	Film	Photo	Run	Scale	Max. error
Lake Pedder	CCN, CCS, CCW	28/01/88	1102	177	65	1:25,000	17.0
	GEC, SAC	28/01/88	1103	45	63	1:25,000	23.9
	MCR, MEW, WEI	28/01/88	1103	89	62	1:25,000	20.8
	MEE	28/01/88	1103	91	62	1:25,000	13.0
	AIE, AIW, EDG	28/01/88	1103	141	67	1:25,000	25.1
Lake St Clair	BCN, BCS, KWW	09/12/93	1208	190	52	1:20,000	20.3
	HAR, KWE, NPE, NPW	10/12/93	1209	67	51W	1:20,000	26.4
	GIT, RCR	10/12/93	1209	76	50	1:20,000	32.9
	FLC	10/12/93	1209	79	50	1:20,000	42.6
	SCN, SCS, TRR	10/12/93	1209	189	49	1:20,000	32.2
	BED	10/12/93	1209	191	49	1:20,000	28.5

Stereoscopic pairs of these photographs as well as additional photographs from different years and/or at different scales were also examined, totalling over 100 photos. The aerial photographs were scanned at a high resolution (1200 dpi) to aid on-screen vegetation delineation and improve subsequent measuring accuracy. The resulting spatial resolution, or pixel size on the ground, was approximately 0.5 m. To minimise positional errors, the scanned photographs were imported into PhotoFactory and orthorectified by registering them to linear ground control features (e.g. road intersections and drainage confluences) derived from 1:25,000 digitised topographic raster maps (Table 2). Control points were registered throughout the photo and at similar altitudes to the study sites in order to minimise potential scale and systematic errors. The maximum positional errors reported are the differences (m) between true and registered locations for the ground control points (GCP) for each photo. The resulting orthophotos and GIS thematic data layers (Table 2) were then imported into ArcView GIS 3.3 with Xtools extension, ArcView 9.0, and ArcEdit (ESRI Inc. 2002), with which all of the subsequent on-screen digitising, editing, classification, geoprocessing, spatial analyses, and map production were conducted.

Table 2. Acquired GIS thematic layers. Positional accuracy is defined as not less than the estimated percentage of well-defined features within x m of true position. Hydline layer includes implied and defined watercourses, hereafter referred to as drainages, and are first-third order streams (after Strahler 1952). All map data projected in Universal Transverse Mercator (UTM), with horizontal Australian Geodetic Datum 1966 Zone 55 (AGD 66).

Title	Format	Scale	Positional accuracy		Attribute accuracy	Source
			(%)	(m)	(%)	
Digital Topographic Maps	Raster	1:25,000	90	17.5	98	TASMAP 2003
Tasmanian Wilderness World Heritage Area Vegetation Mapping (WHAveg)	Vector	1:25,000	N/A	50	Variable	TVMP 2004
Hydline Digital Topographic Series	Vector	1:25,000	90	12.5 -17.5	98	LIST 2003
Transport Segment (Roads)	Vector	1:25,000	90	12.5 -17.5	N/A	LIST 2003
Contours	Vector	1:25,000	90	5	98	LIST 2003
Parks and Wildlife Service Fire History GIS	Vector	1:100,000	N/A	N/A	N/A	PWS Unpublished data; Marsden-Smedley 1998a

Photographic interpretation and digitising

All photographic interpretation and ground-truthing for this study was conducted by the author in order to eliminate likely errors resulting from multiple observers. Initially, site boundaries were visually delineated and digitised in the GIS using fire history information and other site selection criteria (see Chapter 2). Photographic interpretation of vegetation communities within and adjacent to sites consisted of noting apparent differences in tone, texture, size, and shape on the orthophotos (Zonneveld 1988). Personal knowledge of the sites, bird habitat utilisation, and the thematic layers were all used to aid in the location and identification of these features. Stereoscopic pairs of aerial photographs were also examined with a 4x stereoscope in order to aid onscreen delineation by making some features such as drainages, copses, and topographical changes more apparent.

Prior to the commencement of the bird surveys, an intuitive categorisation of habitat features was developed so that all visual observations of birds could be recorded as being in one of seven categories. The primary avian habitat category was the moorland matrix as the moorlands constitute the most extensive and connected vegetation community in the region and were the focus of this study (McGarigal and Marks 1995; Driscoll 2005). The remaining six categories included wood, scrub, and road edges (i.e. lines demarcating boundaries with matrix); drainages (and their associated riparian vegetation zones); and wood and scrub copses. These categories together are referred to as non-matrix habitat. During the course of the bird surveys and after initial assessment of these features on the orthophotos, it was

apparent that the vegetation within the sites and along these edges was often ecotonal in nature (after Risser 1993). This observation is consistent with that reported in the literature (Jarman and Crowden 1978; Balmer 1990). Thus, in some areas floristics and structure were highly variable, even within a relatively fine spatial scale (i.e. metres). For example, although some scrub copses could be identified on the orthophotos and clearly delineated on the ground, others graded almost imperceptibly into the surrounding matrix. Such variability within and ‘fuzzy’ boundaries (Bobbe *et al.* 2001) between vegetation types was difficult to define on the ground and even more difficult to delineate on the orthophotos. Any attempt to consistently and accurately delineate such features across all study sites would have necessitated a rigorous and resource-intensive classification, mapping, and assessment system. Also, in consideration of the large area that would have to be surveyed, it was determined that such an endeavour was beyond the scope of this study. Therefore, the final mapping was based on the four structurally and floristically distinct avian habitat categories, namely the matrix, riparian zones, wood copses, and edges (Figures 1-3); each of which was composed of similar vegetation classes as described below and summarised in Tables 4, 6, and 7.

Vectorised thematic layers were created for each habitat category based on the rasterised orthophotos- polylines for drainage and edge classes, and polygons for the wood copse and matrix classes. Each site was thoroughly examined for these features and was viewed at varying scales in order to locate and identify site boundaries and other habitat features. Most features were delineated and digitised on-screen at a consistent scale (~ 1:2,000) to ensure comparability between sites. The minimum mapping unit for polygons was 0.02 ha, which was small enough to produce a detailed map of the study area and large enough to distinguish most significant habitat features both on the orthophotos and on the ground. It should be noted that many sites contained scattered trees, but these were only mapped as wood copses if they contained discrete clusters of trees ≥ 0.02 ha and were structurally and floristically distinct from the surrounding matrix vegetation. The acquired drainage layer was used as a base for drainages and associated riparian zones; however, some drainage segments were either modified or added if indicated by prior site knowledge and the photographic interpretation. In some cases, sites were adjacent to large drainages channels; however, the actual vegetation bordering the site was often more characteristic of woodland than riparian communities (typically when the distance between the site boundary and the drainage itself was > 25 m). In these cases, the edge habitat was delineated and classified accordingly.



Fig. 1. Riparian zone through eastern moorland matrix habitat (approx. drainage centerline as marked); Beehive Canal North (BCN; 30 years post-fire), Lake St Clair, Tasmania.



Fig. 2. Woodland edge formed with eastern moorland matrix habitat; Flood Creek (FLC; 14 years post-fire), Lake St Clair, Tasmania.



Fig. 3. Wood copse in eastern moorland matrix habitat; Bedlam (BED; 14 years post-fire), Lake St Clair, Tasmania.

Ground-truthing

The primary objective of ground-truthing was to collect reference data to assess the accuracy of the acquired WHAveg classification and drainage layers, and to verify the delineation of site boundaries and wood copses based on the photographic interpretation. In addition, it was necessary to account for possible successional changes in vegetation boundaries, such as between the matrix and woodlands. Ground-truthing included verifying the presence (or absence) of edges, drainages, and copses and obtaining data on their floristics, structure, and location. This process consisted of stratified random sampling (for edges and drainages) and complete sampling for all wood copses. The number of reference samples (i.e. transects, after Skidmore and Turner 1992) was a compromise between the mapping objectives and practicality, and it is recognised that the sample sizes (range = 1-69 per class) fall short of the minimum number of 50 reference samples per vegetation class as recommended by Congalton and Green (1999). Therefore, this limits the statistical inferences that can be made, but does not undermine the utility of the assessment in identifying and correcting problems associated with the original WHAveg mapping scheme (Congalton and Green 1993), particularly as they pertain to the four avian habitat categories (i.e. matrix, riparian, wood copse, edge).

Ground-truthing was conducted March-April 2005. Sampling locations were determined by one of the following methods for each site and were based off of the randomised bird survey transects (see Chapter 4) to help reduce potential biases (e.g. autocorrelation between sample locations; Congalton and Green 1999). For edges, the sampling location was the intersection of closest approach from three randomly selected points at the ends of the transects for each site ($n = 222$); for drainages, it was the intersection of riparian vegetation by the transects ($n = 79$); and for wood copses ($n = 33$), it was the intersection of closest approach from the nearest point along an adjacent transect. At each location a 20 m transect (or longer in the case of wider riparian zones) was then walked perpendicular to the vegetation boundary, such that vegetation in one matrix and one non-matrix polygon were both assessed along approximately 10 m. The actual boundaries between the matrix and non-matrix vegetation were primarily determined by apparent changes in vegetation floristics, structure, and hydrology. The average height of the tallest non-tree stratum was estimated, since no bird observations in the primary tree canopy were recorded for this study (see Chapters 4-6). Heights were measured and assigned to one of four classes (< 1 , 1-2, 2-3, 3-4 m). Floristics were formally evaluated along the transect (and informally at the polygon/line scale) to determine whether they were consistent with the classification codes assigned on the WHAveg map. If the vegetation differed from, or was not included in, the original WHAveg classification (i.e. drainages and wood copses), structurally- and/or cover-dominant species that differentiated these habitats from the matrix were recorded. A GPS (Garmin eTrex, 12 parallel channels, accuracy ± 15 m) was used to record the UTM coordinates of the intersection between the vegetation transect and the habitat boundary. The width of riparian vegetation and the approximate dimensions of wood copses were measured in metres with a laser rangefinder (Leica LRF 800 Rangemaster, accuracy ± 1 m, range 10-800 m) or estimated by pacing (if < 10 m). In some cases features such as drainages or wood copses were identified on the ground but were not mapped, or features such as wood edges were of a different shape and/or location than indicated by the orthophoto map or WHAveg layer. In these cases, the features were either sampled using the above method or were qualitatively assessed and notes were taken accordingly. This was done to ensure that errors of commission and/or omission were minimised at the polygon/line scale, and that the final map would accurately represent features on the ground.

An index of vegetation cover (i.e. density) in matrix habitat was collected for all sites (except AIW and GIT) as part of a collaborative SFT research project on terrestrial invertebrates conducted by the BCB (M. Driessen unpublished data). Cover values were based on ocular estimates of the number of 5 cm^2 squares (out of 200) covered by vegetation on a gridded site board (50 x 100 cm), as outlined in Driessen (1999), and mean values for each site ($n = 7$) were used in subsequent analyses. It is important to note that since sampling was limited

to a relatively small area (~ 0.5 ha) of each site, the cover values may not accurately reflect overall cover across each site.

Vegetation structure analyses

Vegetation height and overall sample sizes were small for many of the separate vegetation classes as well as the wood copse avian habitat category largely due to a lack of availability; therefore, those with similar structure and floristics were pooled into one of three habitat categories (i.e. matrix, riparian zone, edge) for all subsequent analyses. The mean height ($m \pm SE$) of vegetation (excluding mature tree species) for each habitat type was calculated by using the mid-point values of each height class (0.5, 1.5, 2.5, 3.5 m). SPSS 14.0 statistical software (SPSS Inc. 2005) was used to conduct a non-parametric Kruskal-Wallis test to determine whether there were significant differences between mean height class ranks for all habitat categories both within and between Lake Pedder and Lake St Clair. Mann-Whitney U *post hoc* tests with Bonferroni adjusted *P*-values were used to identify any significant differences between categories within each study location. The mean width ($m \pm SE$) of riparian zones for both locations was also calculated, and used to determine the width of cartographic buffers applied to the drainage polylines in order to construct riparian zone polygons and proportional area estimates used in the following GIS spatial analyses.

Regression was used to determine if there were significant relationships between the mean index of vegetation cover for each site and fire age at Lake Pedder and Lake St Clair. An initial examination of scatterplots as well as previous research modelling cover and fuel loading in relation to fire age (Marsden-Smedley and Catchpole 1995b) indicated that there was a curvilinear relationship between cover and fire age. Therefore, curve estimation regression in SPSS 14.0 (SPSS Inc. 2005) was used to test the fit of alternative models. The final regression was based on an S-curve model which provided a plausible fit since the regression curve passed through the origin at zero years post-fire and plateaued at older sites, as predicted based on previous research (see Chapter 1). Model adequacy regarding the assumptions of normality and homogeneity of variance was checked by inspection of residual plots, which did not indicate obvious issues with the models.

Vegetation composition and classification

The results of the ground-truthing were used to revise and update the original WHAveg classification so that it more accurately reflected avian habitat used within the study area. Since vegetation class descriptions have yet to be defined for all of the new amalgamated WHAveg classification codes (S. Corbett pers. comm. 2005), a range of sources were used as the basis for the class descriptions (Jarman *et al.* 1988a; Marsden-Smedley 1993; Driessen 1999; Barnes and Balmer 2000; Balmer and Barnes 2000; Harris and Kitchener 2004; TVMP

2004). It was apparent from the reference data that the flora within the study area differed from some of these class descriptions in terms of both presence, absence, and dominance (structural or cover). Therefore, the reference data were queried in order to identify characteristic species and the class descriptions were modified accordingly. Since most riparian vegetation and wood copses were not mapped as such in WHAveg, the reference data formed the primary basis for these descriptions, with the recognition that these communities shared similarities with some of those described under the various WHAveg classification schemes, as well as by Jarman *et al.* (1988a).

The final classification codes and list of characteristic species identified during ground-truthing were intended to provide a qualitative description of the overall composition and structure of these communities (e.g. sedgeland/heathland vs. woodland), and are not comprehensive species lists derived from rigorous floristic sampling methods. Therefore, these descriptions should be considered accordingly and within the context of this study. A more definitive classification system for buttongrass moorland communities was developed by Jarman *et al.* (1988a) (see Chapter 1) and their comparable classifications are referenced as appropriate herein.

Accuracy assessment

Although there is continuing debate in the remote sensing literature regarding the appropriate methods for conducting an accuracy assessment and presentation of its results, the generation of an error matrix and associated analyses are generally accepted as the standard approach (Foody 2002). Therefore, an accuracy assessment as described by Congalton and Green (1999) was conducted in order to increase the quality of the final habitat map by identifying and correcting WHAveg mapping errors. This was accomplished by comparing the remotely sensed WHAveg classification and habitat delineations with the reference data that were collected during the ground-truthing survey, which are theoretically 100% accurate. A total of 334 reference samples was collected and used for the WHAveg error assessment. The UTM coordinates for the reference samples were mapped in the GIS and used to identify each sample transect and vegetation boundary on the orthophotos. In conjunction with the transect data, these were used to reveal any discrepancies in the classification and shape of WHAveg polygons as compared to those communities and habitat features identified on the ground.

Results of this assessment were recorded in an error (or 'confusion') matrix (after Congalton and Green 1993) which shows the number of polygons assigned to a specific class in the WHAveg classification relative to that determined by the reference dataset ($n = 222$). The diagonal values show when the two classifications are in agreement. This probability, or

overall accuracy (%), was calculated by dividing the sum of these values by the total number of reference polygons. The off-diagonal values show errors of omission or commission by the WHAveg classification. Accordingly, the producer's accuracy is the probability of a reference polygon being correctly classified as such on the map, measuring omission errors; and conversely, the user's accuracy is the probability of a classified polygon being confirmed as such on the ground, measuring commission errors. A separate accuracy assessment was conducted for the wood copses and riparian zones since these were not classified as such in WHAveg. The 33 wood copses were assessed as to whether they were classified as or within woodland polygons in the WHAveg layer, and the 79 drainages as to whether they were included as polylines in the drainage layer.

The values from the error matrix were used to perform a Kappa analysis in program KAPPA, as described by Congalton (1991), which is a discrete multivariate technique used to determine whether the agreement between the classified and reference datasets is statistically significant (assuming a multinomial sampling method). The result of the Kappa analysis is a KHAT statistic (an estimate of Kappa), which is a comparable measure to the overall accuracy reported in the error matrix. The KHAT accuracy represents the difference between the actual agreement (diagonal values) and the row and column totals (of the off-diagonal values), and thus indirectly incorporates errors of omission and commission (Congalton and Green 1999). The Kappa analysis also includes confidence intervals for the KHAT statistic, variance using the Delta method, and a Z statistic, indicating significance at the 95% confidence level (Congalton *et al.* 1983 in Congalton 1991). The accuracy of classification for both site areas and perimeters was determined by calculating the difference in actual area/perimeter measured in the GIS for all classes that were originally used in the WHAveg classification (i.e. not including the riparian and wood copse classes defined for this study) and those on the final map produced for each study site. The sum of the differences was then divided by the totals for all study sites to determine the percentage that was correctly classified in the WHAveg layer (Skidmore and Turner 1992).

Final mapping and spatial analyses

The results from the ground-truthing and accuracy assessment were then used to guide, as appropriate, the modification, addition, deletion, and labelling of habitat features with the revised class descriptions. The resulting final habitat map served as the basis for all the subsequent spatial analyses. Spatial analyses were conducted using standard GIS geoprocessing techniques.

The spatial heterogeneity of landscapes can be measured by its complexity in both composition (i.e. number and proportion of patch types) and configuration (i.e. patch shape,

spatial arrangement, contrast with adjacent patches) (Li and Reynolds 1994; 1995).

Although a plethora of different landscape metrics have been developed over the years, many of these are highly correlated and oftentimes only a small subset of these are necessary to capture a large amount of the spatial heterogeneity in categorical maps (O'Neill *et al.* 1988; Riitters *et al.* 1995; Li and Reynolds *et al.* 1995; Gustafson 1998). The metrics chosen for this study are among those identified in the above studies and are commonly used in ecological research (e.g. McGarigal and McComb 1995; Helzer and Jelinski 1999; Lichstein *et al.* 2002; Kearns *et al.* 2005). The metrics include the following as described by McGarigal and Marks (1995): patch type, area, perimeter, shape index, and edge contrast index. The site polygons were considered to form the habitat edges for the purpose of calculating patch metrics, even though some polygons (e.g. matrix) continued beyond the site boundaries (McGarigal and Marks 1995). Area (ha), perimeter (m), and the proportional abundances (%) of vegetation classes by area and perimeter length were computed for the study area, and are summarised by both locations and sites. Proportional areas of the three habitat categories (i.e. matrix, riparian zone, edge) pooled across vegetation classes were also calculated for each site, although not reported separately herein, and used in subsequent analyses (Chapters 4-5). The edge habitat areas were calculated by creating a 2 m buffer around all edges, including wood copses and disturbed or cleared land (i.e. roadside vegetation dominated by scrub and eucalypt species), as they had similar structure and floristics to the adjacent woodlands and likewise formed distinct edges with the matrix. This area was within the known or assumed fire-affected area and the maximum distance from the matrix edge at which a bird was recorded as within the site. Metrics were calculated using the equations of McGarigal and Marks (1995) as implemented in FRAGSTATS. The shape index is the ratio of perimeter to area normalised to that of a circle; a low value indicates the patch is circular (1.00) while a high value indicates the patch shape is highly irregular. The edge contrast index (%) was calculated by weighting the length of each edge segment (including site boundaries and wood copse perimeters) by the relative structural and floristic contrast between the adjacent patches as derived from the reference dataset. Edge contrast weights (i.e. dissimilarities) ranged from 0.00 for no-contrast boundaries formed by adjacent vegetation patches of the same type (e.g. matrix vs. matrix) to 1.00 for high-contrast 'true' edges formed by vegetation classes that differed markedly in structure and floristics (e.g. matrix vs. woodland). The total edge contrast length for each site was then divided by total perimeter and multiplied by 100 to convert it to a percentage. It is believed that the use of such weighting factors more accurately reflects the functional relationship between different habitat edges and the fauna than treating all edges as functionally equivalent (McGarigal and Marks 1995).

Positional errors

Numerous sources of error may occur at any point during the mapping process, and can contribute to a reduction in overall accuracy of any remote sensing based map. Some of these errors can be controlled for, while others are an inherent part of the process. During the image acquisition process, minor sources of error can arise from lens, film, atmospheric, printing, and scanning distortions; yet, together these may only translate to on-the-ground errors of a couple of metres and are typically ignored in GIS applications. During the orthorectification process, registration errors may also occur, which may arise from human error in registering fiducial and ground control points; however, with care these can be minimised. More significant potential sources of positional errors were calculated for the final habitat map in order to provide information on which to evaluate its accuracy. Relief displacement resulting from photo geometry can be a significant source of error. Displacement increases with increasing radial distance from the photo's principal point and elevation above datum (Wolf 1983). This has been partially controlled for by selecting flat study sites and using photos in which the sites were as close to the principal point as possible. To gain a sense of the scale of such errors for the final habitat map, the maximum horizontal error (at the edge of a given photo) was calculated from the vertical error of the contour layer, and the photo radius and focal length obtained from the specific cameras used for both sets of aerial photos. An additional error may result from digitising; however, since it was conducted on-screen it is difficult to quantify error associated with delineating habitat features. A general rule of thumb is that digitising can be conducted to a tolerance of approximately 0.5 mm, which was multiplied by the photoscale in order to estimate the distance (m) of on-the-ground errors. Finally, accuracy estimates were obtained from the GPS and averaged in order to gain a sense of potential errors associated with the UTM coordinates for the vegetation boundary reference transects.

Results

Vegetation structure, floristics, and classification

The Kruskal-Wallis test indicated that overall mean vegetation heights of the three avian habitat categories were significantly taller at Lake Pedder than at Lake St Clair (Figure 4; $\chi^2 = 34.135$, $df = 1$, $P < 0.001$), largely due to the prominence of shrub species that typify graminoid heathlands. The Kruskal-Wallis test also indicated that there were significant differences in mean ranks of vegetation heights both within Lake Pedder ($\chi^2 = 62.913$, $df = 2$, $P < 0.001$) and Lake St Clair ($\chi^2 = 47.533$, $df = 2$, $P < 0.001$). The results from *post hoc* Mann-Whitney U tests with Bonferroni adjusted $P = 0.025$ (Table 3) indicated that moorland matrix vegetation was significantly shorter than the other habitats as defined. These differences in height and associated structure primarily appeared to be the result of a transition in floristic dominance towards taller shrub (e.g. *Leptospermum* and *Melaleuca* spp.), and to a lesser extent tree species (e.g. *Eucalyptus* spp.) in the non-matrix habitats. A number of these species are the same as those found in the primary matrix classes (i.e. BSW/BSR at Lake Pedder and BEA/ATG at Lake St Clair) as illustrated in the revised WHAveg class descriptions (Table 4). However, some of these appeared to have more robust growth forms in the non-matrix communities. The four new vegetation classes that were developed for this study (Table 4) included Eastern and Southwestern Wood Copses (WCE, WCS) and Eastern and Southwestern Buttongrass Riparian Zones (RIE, RIS).

Results from the curve estimation regression indicated that there was a significant relationship between mean vegetation cover index and fire age at both Lake Pedder ($F = 66.063$; $df = 1, 9$; $P < 0.001$, $R^2 = 0.880$) and Lake St Clair ($F = 67.672$; $df = 1, 11$; $P < 0.001$, $R^2 = 0.860$), with cover increasing and then leveling-off with time since fire (Figure 5). The rate of recovery appeared to be faster in the moderate productivity eastern moorlands at Lake St Clair compared to the low productivity blanket moorlands at Lake Pedder. These findings are in agreement with previous research on cover and fuel loading at these locations that showed a similar trend in post-fire vegetative recovery, such that at comparable ages post-fire, cover and fuel loads were higher in moderate productivity sites than in low productivity sites (Marsden-Smedley and Catchpole 1995b; see Chapter 1).

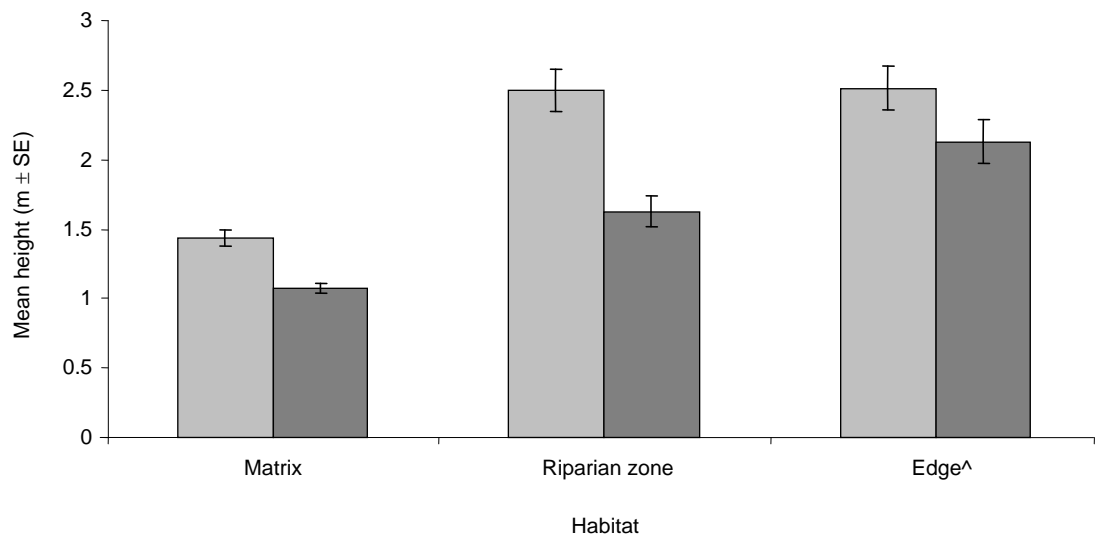


Fig. 4. Mean height (\pm SE) of tallest non-tree vegetation stratum for each habitat category at Lake Pedder (□) and Lake St Clair (■), Tasmania. Means are based on mid-point values of the four height classes (i.e. 0.5, 1.5, 2.5, 3.5 m). ^ Edge habitat includes woodland, wood copse, forest, and roadside vegetation.

Table 3. Comparison of mean ranks of vegetation heights between habitat categories at Lake Pedder and Lake St Clair from *post hoc* Mann-Whitney U tests with Bonferroni adjusted $P = 0.025$.

Location	Habitat category comparison	Z	P
Lake Pedder	Matrix v Riparian zone	-5.818	< 0.001
	v Edge	-5.664	< 0.001
Lake St Clair	Matrix v Riparian zone	-5.936	< 0.001
	v Edge	-7.844	< 0.001

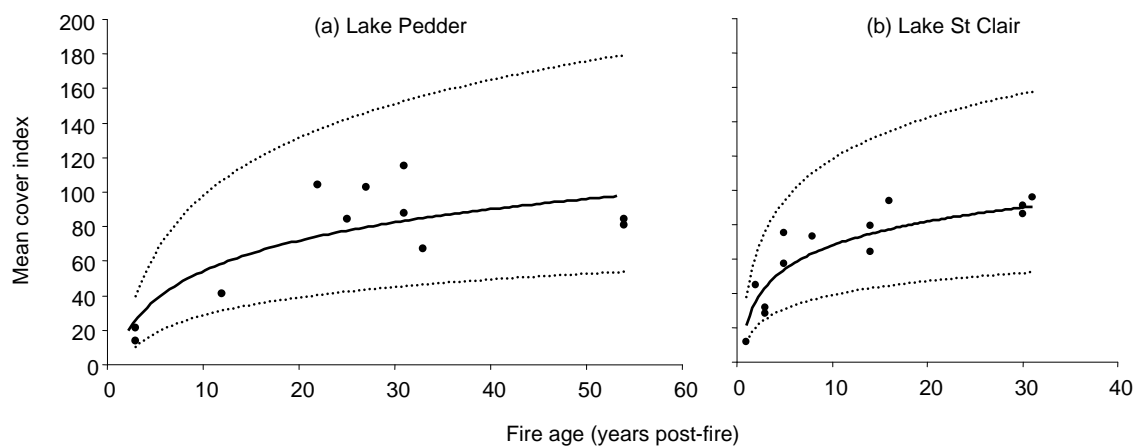


Fig. 5. Mean vegetation cover index (•) versus site fire ages with fitted S-curve regression lines (—) and 95% confidence intervals (.....) at (a) Lake Pedder ($n = 11$) and (b) Lake St Clair ($n = 13$), Tasmania.

Table 4. Vegetation classification legend for the study area. Height and cover data in parentheses are based on the original WHAveg descriptions and not the vegetation data reported above. Classes in *italics* developed for this study. Sources: This study; Jarman *et al.* 1988a; Marsden-Smedley 1993; Driessen 1999; Barnes and Balmer 2000; Balmer and Barnes 2000; Harris and Kitchener 2004; TVMP 2004.

Code	Class Description
ATG	Subalpine Sedgeland – Similar to Eastern Buttongrass (see below), but somewhat more dominated by species typical of wet alpine/subalpine herbfields, including <i>Astelia alpina</i> , <i>Restio australis</i> , <i>Gleichenia alpina</i> , and <i>Empodisma minus</i> . Additional species may include <i>Lepidosperma filiforme</i> , <i>Diplarrena latifolia</i> , <i>Poa</i> spp., <i>Epacris</i> spp. as well as a range of other shrub species. Scattered <i>Eucalyptus amygdalina</i> and <i>E. delegatensis</i> may also be present. Similar to Jarman <i>et al.</i> 's (1988a) Highland Eastern Sedgey (E9), Highland Dry Sedgey (E10), and Pure Buttongrass (E7) communities. Small patches of Sphagnum Bogs (EP3) and Sedgey Fern Bogs (EP4) may also occur.
BEA	Eastern Buttongrass Moorland – (< 2.0 m) Dominated by <i>Gymnoschoenus sphaerocephalus</i> (25-75% cover) and co-dominated by other graminoids including <i>Restio australis</i> , <i>Lepidosperma filiforme</i> , <i>Lepyrodia tasmanica</i> , and <i>Leptocarpus tenax</i> . Associates forming an emergent shrub layer include <i>Sprengelia incarnata</i> , <i>Baeckea gunniana</i> , <i>Epacris gunnii</i> or <i>E. lanuginosa</i> , <i>Boronia citriodora</i> , <i>Almaleea subumbellata</i> , <i>Melaleuca squamea</i> , and <i>M. squarrosa</i> . May also include other species from wet alpine herbfields, such as <i>Gleichenia alpina</i> , <i>Astelia alpina</i> , and <i>Empodisma minus</i> ; and HGR and HHA (see below). May contain scattered <i>Eucalyptus</i> spp.. Similar to Jarman <i>et al.</i> 's (1988a) Layered Eastern Moor (E8) and Common Highland Sedgey (E9a) communities.
BML	Buttongrass Scrubland – <i>Gymnoschoenus sphaerocephalus</i> significantly overtopped by <i>Leptospermum</i> and <i>Melaleuca</i> spp. and sometimes by <i>Banksia marginata</i> and other shrub species. May contain numerous thickets with these species and <i>Eucalyptus nitida</i> . Usually with BSW or BSR affinities (see below), and considered to be later sere between these and WNS.
BSR	Sedgey Buttongrass Moorland – (0.5-1.0 m) Sedgelands low in <i>Gymnoschoenus sphaerocephalus</i> (< 10% cover) but rich in other sedges, particularly <i>Restio</i> spp.. Also <i>Lepyrodia tasmanica</i> , <i>Leptocarpus tenax</i> , <i>Lepidosperma filiforme</i> , <i>Empodisma minus</i> , and <i>Gleichenia dicarpa</i> . Sparse emergent shrub layer or thickets variable in composition, and may include <i>Melaleuca squamea</i> and <i>M. squarrosa</i> , <i>Leptospermum scoparium</i> and <i>L. nitidum</i> , <i>Sprengelia incarnata</i> , <i>Epacris</i> spp., <i>Baeckea leptocaulis</i> , <i>Bauera rubioides</i> , <i>Allocasuarina monilifera</i> , and <i>Boronia</i> spp.. Similar to Jarman <i>et al.</i> 's (1988a) Standard Blanket Moor (B1), Wet Standard (B2), and Southwestern Sedgey (B5) communities.
BSW	Southwestern Buttongrass Moorland – (1.0-1.5 m) Dominated by <i>Gymnoschoenus sphaerocephalus</i> (20-60% cover) and includes other sedges such as <i>Restio</i> spp., <i>Lepyrodia tasmanica</i> , <i>Leptocarpus tenax</i> , <i>Lepidosperma filiforme</i> , and <i>Empodisma minus</i> . Contains a prominent but variable emergent shrub layer (0.6-2.0 m, 10-30% cover) composed of <i>Leptospermum nitidum</i> , <i>Melaleuca squamea</i> , <i>Sprengelia incarnata</i> , <i>Bauera rubioides</i> , <i>Baeckea</i> spp., <i>Boronia</i> spp. and <i>Epacris</i> spp. Wetter areas may form tall (1.0 to > 2.0 m) wet scrub and copses typically dominated by <i>Banksia marginata</i> and <i>Leptospermum</i> and <i>Melaleuca</i> spp., and may contain some of the above species, along with others such as <i>Gahnia grandis</i> and <i>Agastachys odorata</i> . Similar to Jarman <i>et al.</i> 's (1988a) Standard Blanket Moor (B1) and Layered Blanket Moor (B4) communities.
CLL	Disturbed or Cleared Land – Typically found alongside roads and other areas with disturbed and often mounded soils. Variable floristics and structure, primarily composed of species found in adjacent moorlands or woodlands, but usually more robust growth forms and more dominated by shrub and tree species. Species may include <i>Gymnoschoenus sphaerocephalus</i> , <i>Gahnia grandis</i> , <i>Leptospermum</i> and <i>Melaleuca</i> spp., <i>Allocasuarina monilifera</i> , <i>Banksia marginata</i> , <i>Bauera rubioides</i> , <i>Baeckea</i> spp., <i>Epacris</i> spp., <i>Acacia mucronata</i> , and <i>Eucalyptus</i> spp.

Table 4. cont.

Code	Class Description
EAA	<i>Eucalyptus amygdalina</i> Woodland – Dry forests and woodlands often found on rocky dolerite moraines. Dominated by open and uneven-aged <i>E. amygdalina</i> , and may contain <i>E. nitida</i> , <i>E. pauciflora</i> , and <i>E. delegatensis</i> . Understory variable and may contain a wide range of species typical of BEA and HHA.
HGR	Grassland - Tussock grassland dominated by <i>Poa</i> spp. and may also include graminoid species typical of BEA. Similar to Jarman <i>et al.</i> 's (1988a) <i>Poa</i> Grassland (EP1) community.
HHa	Subalpine Heath – (≤ 2.0 m) Variable subalpine to lowland shrubbery dominated by one or more of <i>Pultenaea</i> spp., <i>Oxylobium ellipticum</i> , <i>Cyathodes</i> spp., and <i>Epacris</i> spp. May also contain <i>Leptospermum</i> spp., <i>Callistemon viridiflorus</i> , <i>Banksia marginata</i> , <i>Allocasuarina monilifera</i> , <i>Lomatia tinctoria</i> , <i>Bauera rubioides</i> , <i>Coprosma nitida</i> , <i>Hakea</i> spp., <i>Boronia</i> spp., <i>Poa</i> spp., and <i>Diplarrena latifolia</i> . Often associated with the understory of <i>Eucalyptus</i> woodlands.
HWT	Tall Wet Heath – (1.0-4.0 m) Dominated by <i>Leptospermum</i> spp. and includes one or more of <i>Melaleuca</i> spp., <i>Banksia marginata</i> , <i>Epacris heteronema</i> , <i>Gleichenia dicarpa</i> , <i>Acacia mucronata</i> , and <i>Monotoca</i> spp. Additional species include <i>Bauera rubioides</i> , <i>Gahnia grandis</i> , <i>Agastachys odorata</i> , <i>Cenarrhenes nitida</i> , <i>Boronia</i> spp., <i>Blandfordia punicea</i> , and <i>Richea</i> spp.
LST	Tea Tree Scrub – (≤ 3.0 m) Dominated by <i>Leptospermum</i> spp. May also contain <i>Melaleuca</i> spp., <i>Banksia marginata</i> , <i>Bauera rubioides</i> , <i>Boronia</i> spp., <i>Gymnoschoenus sphaerocephalus</i> , and <i>Eucalyptus</i> spp.
MNF	<i>Eucalyptus nitida</i> Mixed Forest – Forest dominated by <i>E. nitida</i> along with <i>E. delegatensis</i> . Understories are typically composed of thamnic Rainforest species (Note: MNF is included here since BSW vegetation at one site was misclassified as MNF by WHaveg).
MSH	<i>Eucalyptus subcrenulata</i> Mixed Forest – Tall closed forest or subalpine woodlands/thickets dominated by <i>E. subcrenulata</i> . <i>E. johnstonii</i> , <i>E. delegatensis</i> , <i>E. obliqua</i> , and <i>E. coccifera</i> may also be present. Understories are typically composed of thamnic Rainforest species.
RFT	Rainforest - Callidendrous and gallery forests dominated by <i>Nothofagus cunninghamii</i> . May also include <i>Eucryphia lucida</i> , <i>Atherosperma moschatum</i> , <i>Phyllocladus aspleniifolius</i> , <i>Anodopetalum biglandulosum</i> , <i>Richea pandanifolia</i> , and a range of other species.
RIE	Eastern Buttongrass Riparian Zone – Found along drainages. Variable in floristics and structure, often composed of similar species found in adjacent BEA, ATG, and HHA communities; however, usually dominated by sedges that prefer wetter areas and shrub species with more robust growth forms. Typical species include <i>Restio</i> spp., <i>Astelia alpina</i> , <i>Gleichenia</i> spp., <i>Lepidosperma filiforme</i> , <i>Empodisma minus</i> , <i>Diplarrena latifolia</i> , <i>Poa</i> spp., and <i>Gymnoschoenus sphaerocephalus</i> . These species are typically overtopped by one or more of the following shrub species, including <i>Leptospermum</i> spp., <i>Baeckea</i> spp., <i>Epacris</i> spp., <i>Ozothamnus hookeri</i> , <i>Callistemon viridiflorus</i> , <i>Tasmanian lanceolata</i> , <i>Sprengelia incarnata</i> , <i>Melaleuca</i> spp., <i>Olearia obcordata</i> , and a range of other shrub species. <i>Eucalyptus</i> spp. may also be present to varying degrees.
RIS	Southwestern Buttongrass Riparian Zone – Found along drainages. Variable in floristics and structure depending on age and extent. Often composed of similar species found in adjacent BSW, BSR, and related communities. However, usually dominated by shrub species with more robust growth forms, most notably <i>Banksia marginata</i> , as well as <i>Melaleuca</i> spp., <i>Leptospermum</i> spp., <i>Allocasuarina monilifera</i> , <i>Boronia</i> spp., <i>Agastachys odorata</i> , <i>Hakea</i> spp. and <i>Acacia mucronata</i> . Additional species include <i>Gahnia grandis</i> , <i>Bauera rubioides</i> , <i>Calorophus elongatus</i> , <i>Restio</i> spp., <i>Gleichenia</i> spp., as well as some Rainforest species. <i>Eucalyptus</i> spp. are often present to varying degrees. Similar to Jarman <i>et al.</i> 's (1988a) Creek Copses (B13a)

Table 4. cont.

Code	Class Description
SCW	<i>Eucalyptus coccifera</i> Woodland – Dominated by <i>E. coccifera</i> but other <i>Eucalyptus</i> spp. may be present, forming an open subalpine woodland with a HHA understory.
SDH	<i>Eucalyptus delegatensis</i> Woodland – Heathy woodland dominated by <i>E. delegatensis</i> , but may also contain <i>E. pauciflora</i> , <i>E. dalrympleana</i> , <i>E. amygdalina</i> , as well as other <i>Eucalyptus</i> spp. Understory includes species typical of HHA and to a lesser degree BEA.
SLE	<i>Leptocarpus</i> Swamp – Sedgey moorland dominated by <i>Leptocarpus tenax</i> , with <i>Sprengelia incarnata</i> , <i>Gleichenia dicarpa</i> , sometimes <i>Epacris lanuginosa</i> , and <i>Restio</i> spp. May have emergent <i>Hakea</i> spp. and <i>Leptospermum</i> spp. Similar to Jarman et al.'s (1988a) Southwestern Sedgey (B5) community.
SSS	<i>Sphagnum</i> Bogs – Dominated by <i>S. cristatum</i> , but also contains species typical of BEA. Similar to Jarman et al.'s (1988a) <i>Sphagnum</i> Bogs (EP3) community.
WAT	Water – Lakes and impoundments. Edges may contain similar floristics and structure to CLL, BEA, BSW and BSR.
WCE	Eastern Wood Copses – Often similar in floristics and structure to adjacent SDH and EAA. Understories are composed of some BEA and HHA species, typically with more robust growth forms for those shrub species that prefer better drained soils. Some species include <i>Allocasuarina moniflora</i> , <i>Banksia marginata</i> , <i>Leptospermum</i> spp., <i>Acacia verticillata</i> , <i>Lomatia</i> spp., <i>Monotoca linifolia</i> , <i>Epacris</i> spp., <i>Gahnia grandis</i> , and <i>Oxylobium ellipticum</i> .
WCS	Southwestern Wood Copses – Often similar in floristics and structure to adjacent WNF and WNS. Understories are heathy and composed of some BSW species, typically with more robust growth forms for those shrub species that prefer better drained soils. Some species include <i>Banksia marginata</i> , <i>Melaleuca</i> spp., <i>Leptospermum</i> spp., <i>Boronia</i> spp., <i>Bauera rubioides</i> , <i>Gahnia grandis</i> , <i>Agastachys odorata</i> , and <i>Oxylobium ellipticum</i> . Similar to Jarman et al.'s (1988a) Dry Copses (B12b/c) and Wet Copses (B13b/c) communities.
WDF	<i>Eucalyptus delegatensis</i> Wet Forest – Closed tall forest or subalpine woodland dominated by <i>E. delegatensis</i> . Other species include <i>E. pauciflora</i> , <i>E. dalrympleana</i> , <i>E. regnans</i> , <i>E. oblique</i> , and <i>E. coccifera</i> . Understories dominated by <i>Banksia marginata</i> , along with <i>Leptospermum</i> spp. and <i>Melaleuca</i> spp. Additional species may include <i>Bauera rubioides</i> , <i>Gahnia grandis</i> , <i>Boronia citriodora</i> , and <i>Agastachys odorata</i> .
WNF	<i>Eucalyptus nitida</i> Wet Forest – Medium to tall forest dominated by <i>E. nitida</i> and other <i>Eucalyptus</i> spp., including <i>E. subcrenulata</i> and <i>E. delegatensis</i> . Understories are wet shrubberies composed of some thamnisc rainforest species including <i>Leptospermum</i> spp. (any of <i>L. glaucescens</i> , <i>L. scoparium</i> , <i>L. nitidum</i>) and sometimes <i>Phebaleum squameum</i> , <i>Acacia mucronata</i> , <i>Cenarrhenes nitida</i> , <i>Nothofagus cunninghamii</i> , <i>Bauera rubioides</i> , and <i>Gahnia grandis</i> . Also forms tall thickets and creekline scrub with some of the above species, as well as <i>Melaleuca</i> spp., <i>Agastachys odorata</i> , <i>Banksia marginata</i> , <i>Anopterus glandulosus</i> , <i>Phyllocladus aspleniifolius</i> , and <i>Calorophus erostris</i> .
WNS	<i>Eucalyptus nitida</i> Wet Scrub – (3.0-4.0 m) Wet scrub and copses dominated by any of <i>Leptospermum glaucescens</i> , <i>L. scoparium</i> , or <i>L. nitidum</i> , often with a few small emergent <i>Eucalyptus nitida</i> and sometimes other <i>Eucalyptus</i> spp. <i>Acacia mucronata</i> and <i>Banksia marginata</i> may be important, and additional species include <i>Boronia citriodora</i> , <i>Blandfordia punicea</i> , and <i>Agastachys odorata</i> . Wetter areas may have <i>Melaleuca squarrosa</i> , sometimes <i>M. squamea</i> , and <i>Cenarrhenes nitida</i> . Often dense and tangled, with <i>Bauera rubioides</i> , <i>Sprengelia incarnata</i> , <i>Gahnia grandis</i> , and <i>Calorophus erostris</i> .

Accuracy assessment

Table 5 shows the error matrix that quantifies classification differences (i.e. attribute errors) between WHAveg and the reference data collected during ground-truthing. The reference data are theoretically considered to be 100% accurate, but in reality are subject to some errors or differences in interpretation inherent in any classification system, particularly in relation to delineating boundaries between contiguous vegetation communities (Harris and Kitchener 2004). Any such errors may thus contribute to some of those reflected in the error matrix and other accuracy figures reported below for WHAveg.

Two different analyses were conducted to assess the overall accuracy of WHAveg in the study area. The first one was derived from the diagonal values in the error matrix, which indicated where the two datasets agree, and were used to calculate the overall accuracy of 83%. This is just below 85%, which is considered to be the minimal, albeit arbitrary, acceptable level of accuracy for a map product (Congalton and Green 1999). The result from the Kappa analysis, which incorporates off-diagonal values, showed a lower KHAT accuracy of 79% (KHAT 0.79, CI 0.72-0.85, Var. 0.001). Although the Z statistic is typically used to compare two error matrices, the value of 25.0 affirmed, not surprisingly, that the WHAveg classification itself was significantly better than random at the 95% confidence level.

Although these estimates indicated that WHAveg does not meet an acceptable level of overall accuracy in the study area, these figures alone provide little insight into the source of such biases. A closer inspection revealed that most of the off-diagonal values, also reflected in the producer's and user's accuracies, were primarily the result of a few sources of error. The first was between the mapping of matrix classes, such as BEA, and wood edge classes such as SDH. Figure 6 shows the acquired WHAveg and drainage layers superimposed on the orthophoto for an example site, Beehive Canal North (BCN) at Lake St Clair. From this, it was apparent that there was an obvious difference between the two communities. The delineation of their boundaries was subject to a number of factors, including: the scale of mapping, the relative degree of 'smoothing' of irregular edges, the inclusion/exclusion of small vegetation patches (e.g. wood copses) in larger polygons, and placement of boundary lines within ecotonal edges. Therefore, such discrepancies in delineation likely accounted for some of these errors. When they were examined at the polygon scale, it was apparent that most of the vegetation communities themselves were classified correctly and their general polygon shape was similar to that of their corresponding vegetation patches. Yet the polygons were spatially offset from their true position (Figure 6), up to 100 m in some areas. Thus, it was apparent that most of the 'confusion' indicated in the error matrix was not due to misinterpretation between such easily discernable classes, but was largely the result of registration errors. In light of the now somewhat dated method used to map most of the

study area, such errors were not unexpected. The other major source of error seemed to stem from differences in the interpretation and consistent utilisation of some classes. For example, CLL was used to denote vegetation associated with cleared and disturbed land, such as along roads, but had a producer's accuracy of only 7% because it seemed to have been used sparingly, inconsistently, and primarily in the Lake Pedder area. This was despite the fact that in many areas the vegetation along roads not classified as such did show signs of disturbance related to road-building and were structurally and floristically similar to those classified as CLL in WHAveg. Furthermore, some portions along the same stretch of road (i.e. within 2 km) were classified as CLL, while others were not, despite essentially appearing the same on the photos and on-the-ground. When CLL was dropped from the error matrix, then the overall accuracy was increased to 89%. This largely resulted from interpretation differences within WHAveg, perhaps between different staff, and between WHAveg and the classification of reference data, in which all road edges were classified as CLL.

The four new vegetation classes describing wood copses and riparian vegetation for this study (WCE, WCS, RIE, RIS) were not included in the error matrix since they were not explicit classes used in the original WHAveg mapping. Nevertheless, it was still possible to assess the accuracy of the WHAveg and drainage layers in identifying the presence of wood copses and drainages identified in this study. A total of 33 wood copses were identified and sampled for reference data, ranging in size from 0.02-1.31 ha. Of these, the five largest wood copses were identified as woodland vegetation by WHAveg, three as distinct wood copses, and two as parts of larger woodland polygons, for an overall accuracy of only 15%. This low accuracy was largely a reflection of the estimated minimum mapping unit of 0.5 ha in WHAveg. As for the drainages, a total of 79 were traversed and sampled for reference data. Of these, 56 were identified in the drainage layer as either implied or defined watercourses, for an overall accuracy of 71%. This less than acceptable level of accuracy was seemingly the result of a combination of the methodology used to create the original 1:25,000 base maps from which the digital drainage layer was derived, differences in interpretation, and scale.

Table 5. Error matrix comparing the WHAveg classified data to the ground-truthed reference data within the study area.

		Ground-truthed reference data																
Vegetation Class		ATG	BEA	BML	BSR	BSW	CLL	EAA	HGR	HWT	LST	MNF	SCW	SDH	SSS	WAT	Total	User's Accuracy (%)
	ATG	12						2						1			15	80
	BEA		69				1			1				6			77	90
	BML			2		1											3	67
	BSR				18		1										19	95
	BSW					55	11										66	83
	CLL						1										1	100
	EAA		2					3									5	60
	HGR								3								3	100
	HWT																0	0
	LST										3						3	100
	MNF					1											1	0
	SCW		1										1				2	50
	SDH		7											14			21	67
	SSS													1	3		4	75
	WAT					1										1	2	50
	Total	12	79	2	18	58	14	5	3	1	3	0	1	22	3	1	222	
	Producers's Accuracy (%)	100	87	100	100	95	7	60	100	0	100	0	100	64	100	100	Overall Accuracy = 83%	

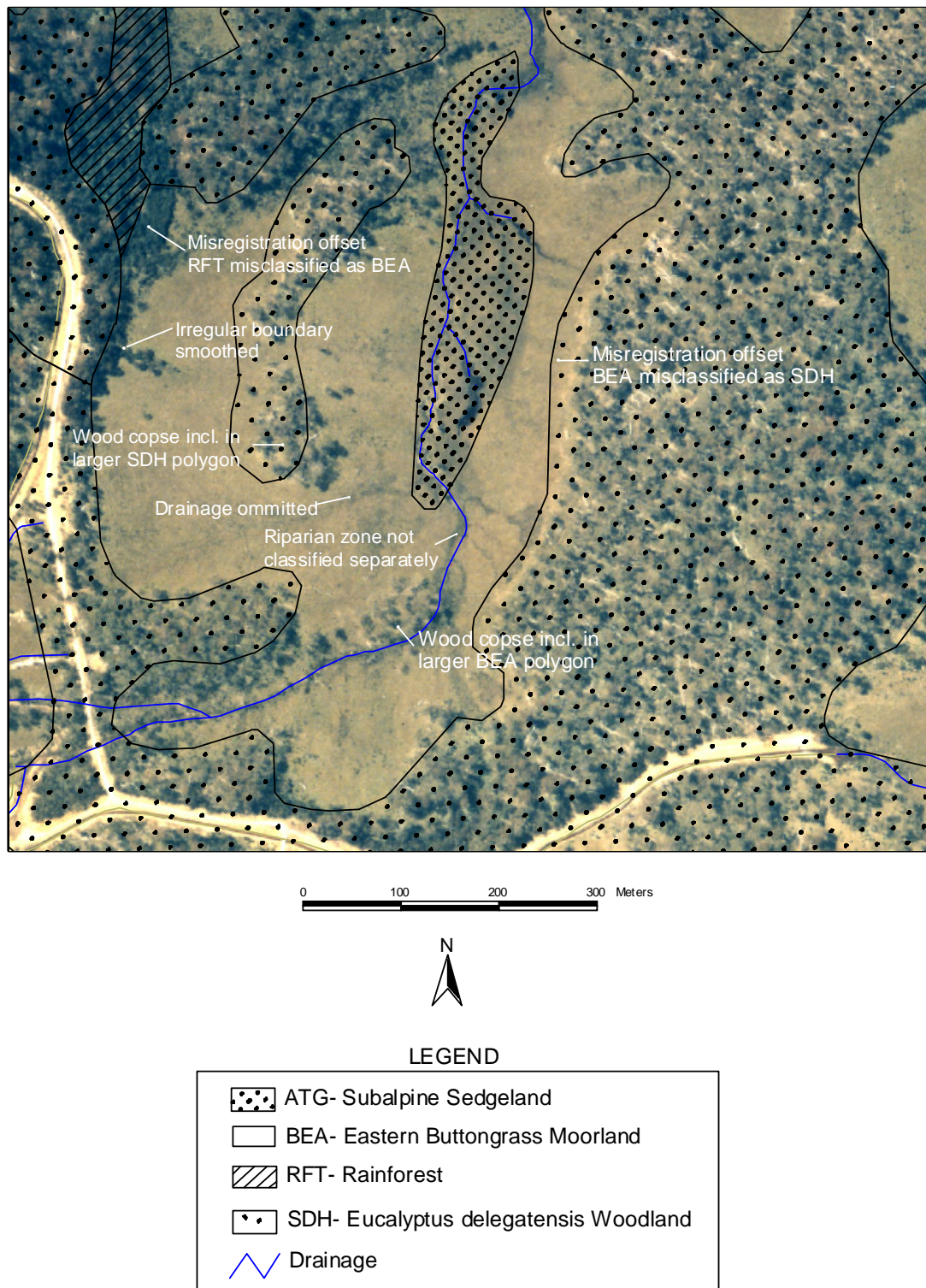


Fig. 6. Acquired WHaveg and drainage layers showing attribute and positional errors in relation to the orthorectified aerial photograph for Beehive Canal North (BCN), Lake St Clair, Tasmania.

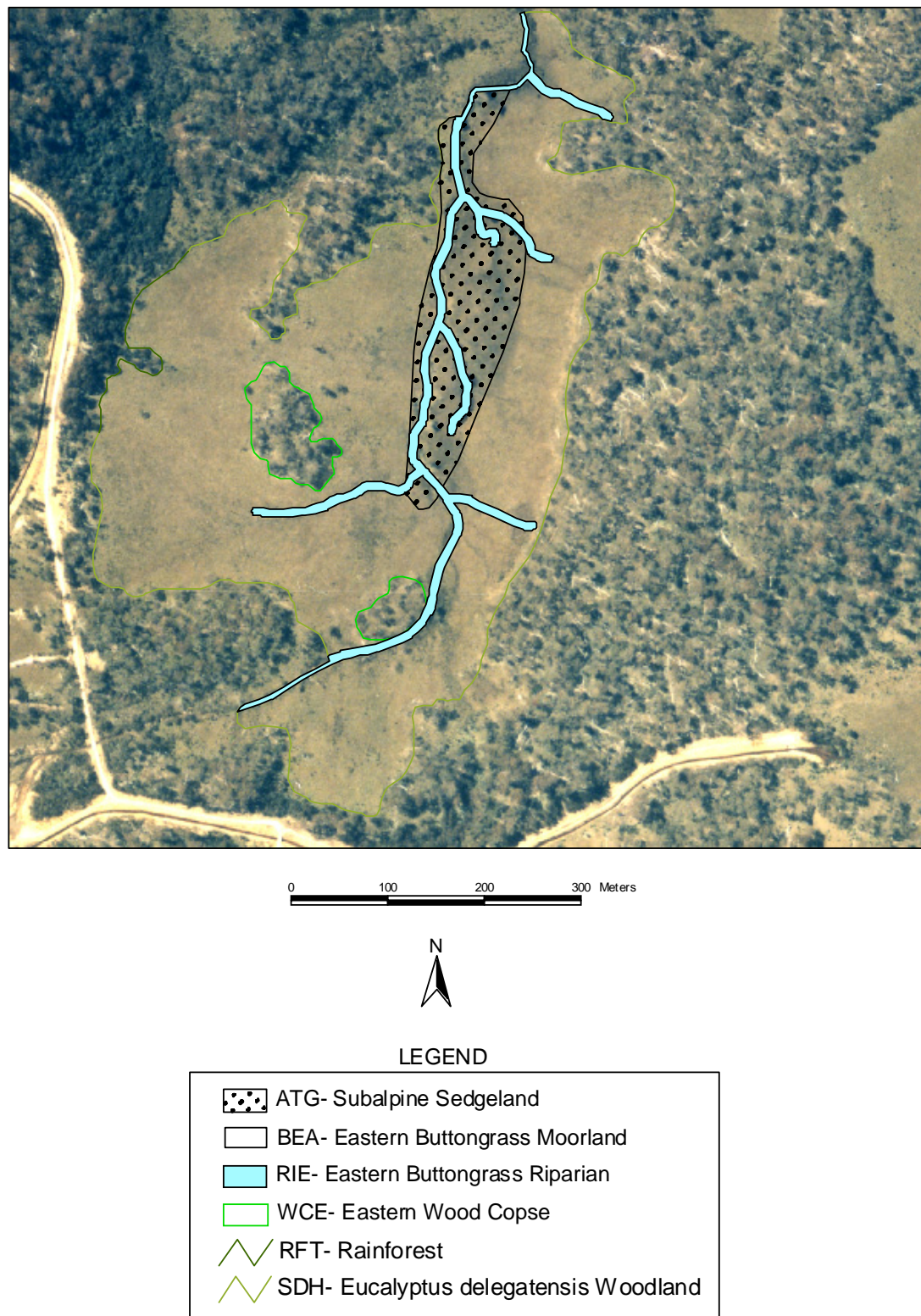


Fig. 7. Final avian habitat map in relation to the orthorectified aerial photograph for Beehive Canal North (BCN), Lake St Clair, Tasmania.

Another issue considered regarding the reported accuracy of WHAveg was temporal differences between the image sources and ground-truthing, both for WHAveg and this study. For example, successional processes may have accounted for some of the differences between classified and reference data. This may be particularly relevant for those sites that have been burnt since the dates of aerial photography, and for those at Lake St Clair since a different run of photos was used herein (1993, 1:20,000) compared to that used for the original WHAveg classification (1988, 1:25,000). However, in consideration of the presence of mature eucalypts in the majority of wood copse and edge habitat (Figure 7), and the relatively slow expected rate of succession (Brown *et al.* 2002), such changes were unlikely to account for the identified scale of offset.

To assess the effects of some of the identified errors in WHAveg on the actual estimates for patch metrics, the percentage of correctly mapped vegetation was calculated for both area and perimeter throughout the study area based on the final area and perimeter figures (Tables 6-7). The accuracy of area estimates from WHAveg clipped by site polygons was only 84%, excluding the RIE, RIS, WCE, and WCS vegetation classes created for this study. It was also clear that WHAveg would have included non-habitat (e.g. woodland classes) within the sites, creating an unacceptable discrepancy between these estimates and the actual habitats surveyed for birds. The accuracy of perimeter estimates was only 6%, although unacceptably low, was expected since boundary accuracy is a conservative measure of map accuracy, meaning relatively small spatial discrepancies can dramatically reduce the estimated accuracy (Skidmore and Turner 1992). Again, most of this error was associated with the offset between the WHAveg polygons and the boundary of vegetation communities as delineated on the orthophotos and confirmed on the ground. Finally, it is also important to note that if one relied upon the WHAveg polygons to help delineate the boundaries of buttongrass moorland patches used in this study, then the subsequent site polygons would have been appreciably displaced and calculated metrics relatively inaccurate. Therefore, within an applied framework the estimated accuracies for WHAveg (i.e. 83% and 79%) were likely biased somewhat high. For reference, the final corrected habitat map for BCN is provided in Figure 7.

Habitat composition and spatial configuration

It was apparent that there was considerable diversity in vegetation classes for both locations based on the percentage of area and perimeter of vegetation classes measured from the final habitat map by study location and site (Tables 6-7). The Lake Pedder area was primarily composed of the two matrix classes, including BSW and BSR. The composition of perimeter was more evenly distributed among the classes, with CLL and RIS contributing the greatest proportions of total edge length. The Lake St Clair area was less diverse; with

the matrix class BEA accounting for the majority of the vegetation. Similarly, the perimeter was dominated by the SDH and RIE classes. It is important to note that 32 drainage segments, totaling 8774 m, were added to the drainage layer composed of the RIS and RIE classes defined specifically for this study. Although this represented a 19% increase in the total drainage length mapped in the study area, the riparian classes covered less than 5% of the study area due to their relatively narrow widths (Figure 7). The wood copse classes as defined (WCE, WCS) only covered approximately 1% of the study area, and were therefore included in the edge habitat category for subsequent analyses. A cursory examination of the area and perimeter figures in Tables 7-8 showed that there were notable differences in many of these estimates both within and between sites of different fire age classes.

The landscape metrics and vegetation cover index for each site were heterogeneous (Table 8). Sites ranged in size from 6.89-83.45 ha and perimeters from 1520-6823 m. Shape indices ranged from 1.16 for SAC that had a roughly circular shape and regular boundaries to 2.63 for KWW that had a non-circular shape and highly irregular boundaries; however, this index was highly sensitive to the site delineation process. Edge contrast indices ranged from 25% for MCR that had only low-contrast moorland edges (i.e. weight = 0.25 for 12 vs. 31 year post-fire matrix) to 100% for GIT that was a small site entirely bound by high-contrast woodland edges (i.e. weight = 1.00 for matrix vs. woodland) (see Chapter 2). Although sampling of cover values was limited to a relatively small proportion of each site, the values obtained for mean cover index were consistent with personal observations of estimated cover within and between each site.

Table 6. Percentage (%) of area by habitat and vegetation class for the study sites.

		Habitat and vegetation class													
	Site	Matrix										Riparian zone		Wood copse	
		ATG	BEA	BML	BSR	BSW	HGR	HWT	LST	SLE	SSS	RIE	RIS	WCE	WCS
Lake Pedder	MEE	0.00	0.00	5.65	0.00	78.07	0.00	0.00	8.29	0.00	0.00	0.00	7.15	0.00	0.85
	MEW	0.00	0.00	0.00	90.49	5.56	0.00	0.00	0.00	0.00	0.00	0.00	3.96	0.00	0.00
	MCR	0.00	0.00	0.00	98.60	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.40	0.00	0.00
	SAC	0.00	0.00	0.00	0.00	94.90	0.00	0.00	0.00	0.00	0.00	0.00	4.70	0.00	0.40
	CCS	0.00	0.00	0.00	0.00	89.61	0.00	0.00	0.00	0.00	0.00	0.00	9.19	0.00	1.20
	CCW	0.00	0.00	0.00	0.00	94.45	0.00	0.00	0.00	0.00	0.00	0.00	5.55	0.00	0.00
	CCN	0.00	0.00	0.00	0.00	87.50	0.00	0.00	0.00	0.00	0.00	0.00	12.50	0.00	0.00
	WEI	0.00	0.00	0.00	90.34	5.08	0.00	0.00	0.00	0.00	0.00	0.00	4.40	0.00	0.19
	GEC	0.00	0.00	0.00	0.00	93.43	0.00	0.00	0.00	0.00	0.00	0.00	5.67	0.00	0.89
	AIE	0.00	0.00	0.00	0.00	98.40	0.00	0.00	0.00	0.00	0.00	0.00	1.60	0.00	0.00
	AIW	0.00	0.00	0.00	0.00	87.69	0.00	2.52	0.00	4.48	0.00	0.00	5.32	0.00	0.00
	EDG	0.00	0.00	0.00	0.00	95.83	0.00	0.00	0.00	0.00	0.00	0.00	4.17	0.00	0.00
Lake St Clair	HAR	0.00	96.65	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.35	0.00	0.00	0.00
	SCN	98.44	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.56	0.00
	KWE	0.00	94.59	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.41	0.00	0.00	0.00
	NPW	0.00	87.90	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.38	0.00	7.72	0.00
	BCS	0.00	82.30	0.00	0.00	0.00	12.03	0.00	0.00	0.00	0.00	4.15	0.00	1.52	0.00
	KWW	0.00	96.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.78	0.00	0.00	0.00
	SCS	89.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	8.57	0.00	2.34	0.00
	BED	0.00	91.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.58	0.00	2.25	0.00
	FLC	0.00	97.80	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.20	0.00	0.00	0.00
	NPE	0.00	91.98	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.53	0.00	2.49	0.00
	BCN	9.76	79.99	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.66	0.00	4.58	0.00
	RCR	0.00	99.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.95	0.00	0.00	0.00
	TRR	0.00	90.70	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.84	4.58	0.00	0.88	0.00
	GIT	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 7. Percentage (%) of perimeter length (i.e. edge) by vegetation class for the study sites (excluding wood copses). Edge contrast weights relative to matrix, used to calculate edge contrast index.

		Vegetation class (Edge contrast weights)																				
		ATG (0.00)	BEA (0.00)	BML (0.50)	BSR (0.25)	BSW (0.00)	CLL (0.25- 0.50)	EAA (1.00)	HHA (0.75)	HWT (0.75)	LST (0.75)	MSH (1.00)	RIE (0.50)	RIS (0.50)	RFT (1.00)	SCW (1.00)	SDH (1.00)	SSS (0.00)	WAT (0.00)	WDF (1.00)	WNF (1.00)	WNS (1.00)
	Site																					
Lake Pedder	MEE	0.00	0.00	23.72	0.00	10.87	15.24	0.00	0.00	0.00	41.07	0.00	0.00	8.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.79
	MEW	0.00	0.00	0.00	21.31	0.00	18.18	0.00	0.00	0.00	0.00	0.00	0.00	18.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	42.32
	MCR	0.00	0.00	0.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	SAC	0.00	0.00	0.00	0.00	47.66	34.38	0.00	0.00	0.00	0.00	0.00	0.00	15.84	0.00	0.00	0.00	0.00	0.00	2.13	0.00	0.00
	CCS	0.00	0.00	0.00	0.00	11.14	42.83	0.00	0.00	0.00	0.00	0.00	0.00	23.86	0.00	0.00	0.00	0.00	0.00	0.00	0.00	22.16
	CCW	0.00	0.00	0.00	0.00	31.54	27.89	0.00	0.00	0.00	0.00	0.00	0.00	36.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	4.56
	CCN	0.00	0.00	0.00	0.00	8.28	19.51	0.00	0.00	0.00	0.00	0.00	0.00	72.21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	WEI	0.00	0.00	0.00	30.46	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	9.05	0.00	0.00	0.00	0.00	0.00	0.00	33.24	27.25
	GEC	0.00	0.00	0.00	0.00	15.18	32.42	0.00	0.00	0.00	0.00	0.00	0.00	32.59	0.00	0.00	0.00	0.00	0.00	0.00	0.00	19.81
	AIE	0.00	0.00	0.00	0.00	20.28	57.10	0.00	0.00	0.00	0.00	0.00	0.00	10.62	0.00	0.00	0.00	0.00	7.15	0.00	0.00	4.85
	AIW	0.00	0.00	0.00	0.00	8.60	51.59	0.00	0.00	1.24	0.00	0.00	0.00	32.86	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.71
	EDG	0.00	0.00	0.00	0.00	19.80	40.13	0.00	0.00	0.00	0.00	0.00	0.00	30.84	0.00	0.00	0.00	0.00	0.00	0.00	9.23	0.00
Lake St Clair	HAR	0.00	7.88	0.00	0.00	0.00	6.42	0.00	0.00	0.00	0.00	0.00	7.59	0.00	0.00	0.00	78.12	0.00	0.00	0.00	0.00	0.00
	SCN	10.34	0.00	0.00	0.00	0.00	0.00	85.88	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.78	0.00	0.00	0.00	0.00	0.00
	KWE	0.00	1.63	0.00	0.00	0.00	4.20	0.00	4.59	0.00	0.00	0.00	50.39	0.00	0.00	0.00	39.19	0.00	0.00	0.00	0.00	0.00
	NPW	0.00	18.50	0.00	0.00	0.00	22.43	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	59.07	0.00	0.00	0.00	0.00	0.00
	BCS	0.00	10.87	0.00	0.00	0.00	5.03	0.00	7.25	0.00	0.00	2.84	20.39	0.00	11.06	0.00	34.25	0.00	0.00	0.00	8.31	0.00
	KWW	0.00	2.63	0.00	0.00	0.00	3.73	0.00	0.00	4.82	0.00	0.00	22.17	0.00	0.00	0.00	66.66	0.00	0.00	0.00	0.00	0.00
	SCS	0.00	0.00	0.00	0.00	0.00	0.00	22.31	0.00	0.00	0.00	0.00	32.87	0.00	0.00	0.00	44.82	0.00	0.00	0.00	0.00	0.00
	BED	0.00	1.71	0.00	0.00	0.00	0.00	41.96	0.00	0.00	0.00	0.00	40.80	0.00	0.00	0.00	15.53	0.00	0.00	0.00	0.00	0.00
	FLC	0.00	0.00	0.00	0.00	0.00	6.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	93.66	0.00	0.00	0.00	0.00	0.00
	NPE	0.00	4.07	0.00	0.00	0.00	2.87	0.00	0.00	0.00	0.00	0.00	31.95	0.00	0.00	0.00	61.10	0.00	0.00	0.00	0.00	0.00
	BCN	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	8.17	0.00	9.88	0.00	81.95	0.00	0.00	0.00	0.00	0.00
	RCR	0.00	0.00	0.00	0.00	0.00	13.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	86.81	0.00	0.00	0.00	0.00	0.00
	TRR	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	35.97	0.00	0.00	0.00	58.86	5.17	0.00	0.00	0.00	0.00
	GIT	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	27.17	72.83	0.00	0.00	0.00	0.00	0.00

Table 8. Landscape configuration metrics and vegetation cover for the study sites. *Actual cover not measured by BCB; therefore, based on personal observations it was considered to be the mean value of the adjacent 54 year old sites (AIE and EDG) so that AIW could be included in subsequent analyses.

	Site	Area (ha)	Perimeter (m)	Shape index	Edge contrast index (%)	Mean cover index
Lake Pedder	MEE	53.34	5943	2.30	59.56	14.29
	MEW	48.94	3927	1.58	61.29	20.71
	MCR	21.22	2748	1.68	25.00	41.07
	SAC	31.48	2300	1.16	34.16	103.57
	CCS	33.06	3098	1.52	54.39	84.29
	CCW	23.92	2525	1.46	36.45	102.86
	CCN	15.18	1872	1.36	40.96	115.00
	WEI	38.77	3460	1.57	66.05	87.86
	GEC	20.20	2377	1.49	57.58	66.57
	AIE	13.62	1957	1.50	38.74	83.57
	AIW	11.75	1696	1.40	48.91	82.14*
	EDG	13.12	1875	1.46	44.69	80.71
Lake St Clair	HAR	32.65	3432	1.69	83.73	12.14
	SCN	7.61	1799	1.84	90.49	45.00
	KWE	22.48	3077	1.83	68.78	32.14
	NPW	16.98	2559	1.75	70.72	27.86
	BCS	83.45	6823	2.11	77.36	57.14
	KWW	45.39	6278	2.63	83.25	75.00
	SCS	21.67	4317	2.62	85.16	73.21
	BED	19.79	3091	1.96	81.46	63.57
	FLC	47.68	5554	2.27	95.41	79.29
	NPE	70.58	6304	2.12	80.54	94.29
	BCN	23.72	3209	1.86	96.54	86.43
	RCR	18.95	2699	1.75	90.10	91.43
	TRR	39.04	3738	1.69	78.45	96.43
	GIT	6.89	1520	1.63	100.00	—

Positional errors

Perhaps the greatest potential sources of estimated positional errors of the final habitat map (Table 9) during orthorectification arose from the (in)accuracy of the base layers, including the acquired drainage and road layers used for visual registration and the contour layer used as the base for the digital elevation model. The acquired layers can vary from ≤ 12.5 -17.5 m horizontal displacement depending on the specific image processing method used to map a given feature, and ≤ 5 m vertical displacement. Based on the latter, and photo radius and focal length of the cameras used for both sets of aerial photos, an additional error of 1.9-3.8 m may have occurred at sites that were located near the edge of the orthorectified photos. Since photos were chosen in which sites were located as close to the photo's principal point as possible, the amount of error contributed from such distortions were likely to be less than these figures indicate. Both of these potential sources of horizontal displacements were largely reflected in the maximum positional errors reported in Table 1 for each orthorectified

photo, and their means of 20.0 and 30.5 m reported below for Lake Pedder and Lake St Clair, respectively.

Table 9. Summary of estimated on-the-ground positional errors for final habitat map (\pm m) ^ (LIST 2003; TASMAP 2003).

Location	Acquired base layers^	Max. at photo edge	Mean of max. for orthophotos	Digitising	Mean GPS location
Lake Pedder	12.5-17.5	3.8	20.0	12.5	6.1
Lake St Clair	12.5-17.5	1.9	30.5	10.0	6.1

The digitising process may have also contributed errors ranging from 10.0-12.5 m based on the 0.5 mm digitising rule of thumb; however, these estimates were difficult to separate from the inherent subjectivity of delineating site boundaries that were often of an ecotonal, or fuzzy, nature. For example, Balmer (1990) found that the transition zone between true forest and true moorland at one site was approximately 10 m. Furthermore, the GPS used to determine the UTM coordinates of vegetation boundaries had a mean error of 6.1 m which may have also contributed to discrepancies between the UTM points and apparent boundaries on the orthophotos. In such cases, boundaries for the final habitat map were delineated so that they were consistent with on the ground observations. Skidmore and Turner (1992) conceptualised such boundary errors associated with rectification, map compilation, and ecotone width as making up an ‘error zone’ within which it is unclear as to which of two adjacent polygons a given pixel should be classified. Further investigation of and possible correction for these error zones was beyond the scope of this study.

Discussion

The use of digital maps developed from remote sensing and GIS techniques has become increasingly common in wildlife research, particularly for studying habitat associations at the landscape-scale. As more map products are being developed for a range of land management applications, it is often possible to acquire some of these ‘off-the-shelf’ layers. These can either be used ‘as is’, or as a base for defining, mapping, and quantifying project-specific habitat features. Unfortunately though, in many cases researchers do not adequately assess and report the potential limitations of these map products or their derivatives on their subsequent results (Glenn and Ripple 2004).

To describe and quantify habitat utilised by birds for this study, the WHAveg GIS layer was acquired since it provided a detailed classification of vegetation throughout the study area. The overall accuracy results from the error matrix (83%) and Kappa analysis (79%) indicated that the WHAveg classification within the study area did not meet the generally

accepted accuracy standard of 85%. Additional analyses also indicated that important avian habitat features, such as wood copses and riparian zones, were either inconsistently classified or omitted altogether since they were not considered as part of their overall mapping objectives, were not readily identified from aerial photos, or were below the minimal mapping unit (~ 0.5 ha within the study area). These low accuracy estimates were largely the result of a range of interpretation, classification, delineation, and registration errors that are common to many mapping projects (Congalton and Green 1993). Since positional errors such as those caused by misregistration tend to confound the sources of any attribute errors, it is difficult to determine the extent to which each of these contributes to the overall accuracy (Foody 2002). It has been shown that misclassification alone can cause significant bias in area and perimeter estimates (Czaplewski 1992), and the accuracy estimates of 84% and 6%, respectively, for WHAveg bring this to bear. Furthermore, the magnitude of the underlying errors, such as the inclusion of non-habitat (e.g. woodland classes) within the study sites, must be considered as well. Since the utility of a map is a function of its accuracy relative to the scale and objectives of a study (Foody 2002), and the main objective here was to classify and accurately quantify all avian habitat features in the study area, it was apparent that the WHAveg and associated layers could not simply be used 'as is'. Thus, the results from the accuracy assessment were used to correct some of these errors to the extent possible, and used as a base for creating a finer-scale and greater-resolution classified bird habitat map that was more inclusive, accurate, and appropriate for the aims of this study.

As is the case with any map, there were numerous potential sources of attribute and positional errors associated with the final habitat map produced for this study. However, the quantification of such errors is considered to be a complex task, even for studies focused on mapping in and of itself (Foody 2005). This is exacerbated by the fact that these errors tend to accumulate through the mapping process (Congalton and Green 1999). Despite this being the case, it is possible to provide some overall accuracy estimates in relation to those reported for WHAveg and the other acquired layers. In consideration of the relative scale and degree of ground-truthing conducted for this study, it is a reasonable assumption that attribute accuracy was markedly higher than that estimated for WHAveg, but no greater than that reported for the other base layers (e.g. drainages), and thus was likely close to 98%. Positional accuracy is somewhat harder to quantify into one figure since numerous sources of error may be at play and a more comprehensive assessment would have required methods (e.g. surveying) that were beyond the scope of this study. Since the accuracy of a map cannot surpass that of the base layers on which it is based (if uncorrected), it can be assumed that the best case scenario was positional accuracies around 12.5 m (based on lowest reported error for drainage and road layers) and a worse case scenario around 43 m (based on largest maximum horizontal error for the orthophotos). In either case, these errors were less

than the estimated positional accuracy of 50 m reported in the WHAveg metadata, and considerably less than the actual maximum offsets (~ 100 m) measured for WHAveg in this accuracy assessment. Therefore, in relation to the scale and objectives of this study, this level of positional accuracy was considered to be acceptable. Moreover, since the same image sources and methods were used for all sites within the study locations, the relative accuracy was adequately high. The mapping methods used herein have ensured that the final habitat map was both logically consistent (i.e. all features are labelled with only one class and all polygons are closed) and complete (i.e. all sites were mapped in their entirety).

The results of this fine-scale mapping project demonstrated that there was a notable improvement in both attribute and positional accuracies in the final habitat map, thus minimising potential biases in the calculated habitat metrics and the possibility of spurious avian habitat associations in subsequent analyses presented in Chapters 4-6. The process necessary to thoroughly assess existing GIS layers and to modify and amend them accordingly was both resource and time consuming. Thus, the development of project-specific map products may not necessarily be worthwhile for all such wildlife research projects and must be evaluated against the relative scales (i.e. mapping and ecological) and objectives of a given study.

The significant differences identified in vegetation heights between the avian habitat categories (i.e. matrix, riparian zone, and edge) and their characteristic floristics helped to substantiate their *a priori* classification. These findings are in agreement with previous research in blanket moorlands (Marsden-Smedley 1990; Brown and Podger 1982), and are likely the result of interspecific competition that may be attributed to minor changes in drainage as well as peat type (e.g. from muck to fibrous peat). These differences are also manifested in their fire sensitivity and flammability attributes, with moorlands classified as having low sensitivity and very high flammability, while the edge-forming wet sclerophyll woodlands and forests have a high fire sensitivity and moderate flammability (Pyrke and Marsden-Smedley 2005). In addition, the significant relationship between mean index cover and time since fire is consistent with both previous research and current theory on post-fire succession in Tasmanian buttongrass moorlands (see Chapter 1). However, results from the vegetation analyses showed that despite attempts to minimise other abiotic and biotic variables among the study sites as described in Chapter 2, there were notable differences in vegetation composition, structure, and spatial configuration both within and between study sites. Inter-site differences in landscape metrics are largely the result of varying, complex, and long-term interactions between vegetation and factors such as soil conditions, hydrology, and fire (Jarman *et al.* 1988a; Marsden-Smedley 1990; Balmer 1991). However, these differences were also partially artifacts of the site selection and delineation process (see

Chapter 2). The relative influence of such spatial heterogeneity, in addition to fire age *per se*, on avian habitat use, composition, and densities is explored in Chapters 4-6. Finally, this avian habitat map, along with the results in subsequent chapters, may be used as a foundation for the potential future development of landscape-scale bird-habitat prediction models in buttongrass moorlands (see Chapter 7).

Avifaunal composition and densities in relation to post-fire succession of buttongrass moorlands in the Tasmanian Wilderness World Heritage Area

Introduction

Fire is perhaps the principal abiotic agent in Australian ecosystems (Jackson 1968; Bowman 2000; Clark *et al.* 2002) and avifaunal responses to disturbance by fire are complex, highly variable, and species- and context-specific (for reviews see Woinarski and Recher 1997; Woinarski 1999a, 1999b; Chapter 1). Paradoxically, many Australian vegetation communities including sedgelands and heathlands are characterised by being both highly flammable throughout much of the year (Specht 1979a; Keith *et al.* 2002a; Pyrke and Marsden-Smedley 2005) and supporting resident bird species that possess a number of attributes associated with fire-sensitivity, such as being ground-dwelling and cover-dependent, and having limited dispersal capabilities and low fecundity (Kikkawa *et al.* 1979; Baker 2000; Woinarski and Recher 1997; Burbidge *et al.* 2005). Since fires in heathlands often burn much of the above-ground vegetation, species that exhibit one or more of these attributes may be less likely to survive the fire, find adequate resources in the post-fire environment, evade predators, recolonise from adjacent unburnt areas, or successfully reproduce (Fox 1978; Baker 2000, 2002; Keith *et al.* 2002a; Whelan *et al.* 2002). The rate and extent of the recovery of vegetation after fire may influence the relative suitability of habitats for both resident and opportunistic species through changes in vegetation structure and resource availability over time (Kikkawa *et al.* 1979; Brown 1991; Keith *et al.* 2002a; Whelan *et al.* 2002). Numerous studies in Australian heathlands have demonstrated that bird species composition and abundance may be affected by fire (e.g. Smith 1987; McFarland 1988b, 1994; Brooker and Rowley 1991; Recher 2005). However, bird populations do not necessarily follow a predictable successional pathway and may be influenced by a range of factors other than, or in addition to, habitat structure and fire age *per se* (Meredith *et al.* 1984; Woinarski and Recher 1997; Baker 2002). Although many Australian ecosystems and their fauna are considered to exhibit adaptations to fire disturbance (Catling and Newsome 1981; Recher and Christensen 1981), changes in fire regimes (e.g. time since fire, season, patchiness, extent, intensity, frequency) since European colonisation and the resulting direct or indirect mortality have either been confirmed or implicated as contributing to the extinction of at least two species and three subspecies of Australian birds (Woinarski and Recher 1997). Currently, fire regimes that are outside of their historical range of variation are a threat to at least 51 bird taxa, including many heathland species (Garnett 1992;

Woinarski 1999a, 1999b; Garnett and Crowley 2002; Olsen and Weston 2005). Short inter-fire intervals and increased fire frequency are considered to be the major threats to many bird species, particularly for mid- to late-successional heathland species that cannot persist and reproduce in early-successional habitats (Brooker and Rowley 1991; Mushinsky and Gibson 1991; Woinarski and Recher 1997). These circumstances present a major challenge for fire and conservation managers. It is generally recognised due to the extremely wide range of observed avian responses to fire and the complexity of the underlying processes that sound decision-making is contingent upon detailed, ecosystem-specific research (Wilson 1994; Woinarski and Recher 1997; Baker 2002; Whelan *et al.* 2002; Paton *et al.* 2005; Recher 2005; Tasker and Baker 2005).

Buttongrass moorlands form an ecosystem that exemplifies the complex interplay of fire, soils, flora, and avifauna. They are comprised of sedgeland and graminoid heathland communities typically dominated by the hummock-forming tussock sedge commonly named buttongrass (*Gymnoschoenus sphaerocephalus*) (Specht 1979a; Jarman *et al.* 1988a). Buttongrass moorlands cover large areas in Tasmania (0.55 million ha) and are most extensive in the perhumid, oligotrophic peatlands of western Tasmania where they are largely protected within the Tasmanian Wilderness World Heritage Area (TWWHA; 335,000 ha) (Brown *et al.* 1993; Smith and Banks 1993; Balmer *et al.* 2004; TVMP 2004). Buttongrass moorlands are highly pyrogenic, but are adapted to and have a low sensitivity to fire, and exhibit post-fire changes in vegetation structure and composition that are characteristic of secondary succession (Connell and Slatyer 1977; Brown *et al.* 2002; Balmer *et al.* 2004; Pyrke and Marsden-Smedley 2005; see Chapter 1).

To date, knowledge of the Tasmanian moorland avifauna has been very limited and is primarily based on qualitative descriptions (e.g. Gellie 1980; Brown *et al.* 1993; Driessen 2006). However, all sources report that the avifauna is depauperate and is comprised of only three resident species that are known to depend exclusively upon moorlands in the study area, namely the Southern Emu-wren (*Stipiturus malachurus littleri*), Striated Fieldwren (*Calamanthus fuliginosus diemenensis*), and Ground Parrot (*Pezoporus wallicus wallicus*), and a small number of transient species that are typically associated with adjacent woodlands, forests, and related habitats (Brown *et al.* 1993; see Chapter 1). Bryant's research (1991, 1992, 1994) on the density, distribution, and conservation status of the Ground Parrot is the only major work to date that has focused on a moorland resident species in Tasmania; however, its scope in relation to fire ecology was limited and it did not investigate any other members of the avifauna (S. Bryant pers. comm. 2003). Otherwise, our knowledge of the resident species is limited to either old observational studies conducted in non-moorland habitat in other regions of the State (e.g. Legge 1908; Fletcher 1913a,

1913b, 1915a; Lord 1927; Sharland 1953), or on more recent studies primarily on different subspecies on the Australian mainland (e.g. Gosper and Baker 1997; Burbidge *et al.* 2005; Maguire 2006a, 2006b). Although extensive research investigating the effects of fire on the Ground Parrot has been conducted on the mainland, results have varied and their implications for fire management have been equivocal (Meredith *et al.* 1984; McFarland 1991a, 1991b, 1991c; Woinarski 1999a, 1999b; Baker 2002; Tasker and Baker 2005; Burbidge *et al.* 2007). To date, no detailed fire ecology studies either on the mainland or in Tasmania have focused on the Southern Emu-wren or Striated Fieldwren (Woinarski 1999a, 1999b; Higgins *et al.* 2001; Higgins and Peter 2002). Accordingly, over the years a number of researchers have identified the need to study the Tasmanian moorland avifauna, particularly in relation to the effects of fire, in order to help guide fire management and conservation efforts (e.g. Gellie 1980, Eberhard 1987; Bryant 1991; Driessen 2001).

Despite the lack of prior research on the Tasmanian resident avifauna, it is evident from the literature that the Southern Emu-wren, Striated Fieldwren, and Ground Parrot all exhibit some fire-sensitive attributes since they are habitat specialists, cryptic, ground-dwelling, dependent on dense cover, and except for the latter, are poor fliers and are considered to have limited dispersal capabilities (Meredith *et al.* 1984; Bryant 1990; Gosper and Baker 1997; Pickett 2000; Higgins and Peter 2002; Wilson and Paton 2004; Tasker and Baker 2005; Burbidge *et al.* 2007; see Chapter 1). Accordingly, extensive wildfires in increasingly fragmented habitats, short inter-fire intervals, high fire frequencies, and otherwise modified fire regimes have been implicated in the local extinction of some mainland populations, and have been identified to be among the primary threats to extant populations (MLRSERT 1998; Garnett 1992; Woinarski 1999a; Garnett and Crowley 2000; Lunney *et al.* 2000). In part due to such fire-induced impacts, five mainland subspecies of Southern Emu-wren are listed as being threatened or endangered (*S. m. hartogi*, Western Australia *Wildlife Conservation Act of 1950*; *S. m. intermedius*, *S. m. parimeda*, *S. m. halmaturinus*, and *S. m. malachurus*, South Australia *National Parks and Wildlife Act 1972* and *Environment Protection and Biodiversity Conservation Act 1999*); a mainland subspecies of Striated Fieldwren is listed as vulnerable (*C. f. albiloris*, New South Wales *Threatened Species Conservation Act 1995*); and both subspecies of Ground Parrot are listed on the mainland as either vulnerable (*P. w. wallicus*, Queensland *Nature Conservation Act 1992* and New South Wales *Threatened Species Conservation Act 1995*) or endangered (*P. w. flaviventris*, Western Australia *Wildlife Conservation Act of 1950* and *Environment Protection and Biodiversity Conservation Act 1999*) (Garnett and Crowley 2000). None of the resident species are currently listed in Tasmania; however, Ground Parrot populations have declined and some populations have become locally extinct, particularly in northern and eastern regions where suitable habitat was historically limited (Bryant 1991). Despite the historical

and ongoing threats posed by development, grazing, and increased fire frequency in some parts of Tasmania, populations are considered to be secure in the southwest where they are free from most threats under the protection of the Tasmanian *Nature Conservation Act 2002* and the TWWHA (Bryant 1991). It has also been suggested that inappropriate fire regimes may be a threat to populations of the Southern Emu-wren and Striated Fieldwren in Tasmania (Legge 1908; Gellie 1980).

Currently, there is considerable debate concerning the most appropriate way to manage fire within the TWWHA in order to protect life and property, and conserve its biodiversity, particularly within buttongrass moorland ecosystems (PWS 1999; DPIW 2007; May and Balmer 2008). Since inappropriate fire regimes are a demonstrated or proposed threat to mainland populations of the Southern Emu-wren, Striated Fieldwren, and Ground Parrot, as well as a number of other species known to use moorlands (Garnett 1992; Brown *et al.* 1993; Garnett and Crowley 2000; see Chapter 1), it is imperative to investigate the ecological attributes and responses of the moorland avifauna to fire in order to help guide management across Tasmania. The research described here is the first community-level study of the buttongrass moorland avifauna and was primarily focused on exploring the influence of site fire age and productivity on the avifauna, since these are the primary variables in the fire behaviour and prediction models currently used by the Parks and Wildlife Service to guide prescribed burning and wildfire suppression operations in the TWWHA (Marsden-Smedley *et al.* 1999; PWS 2004; Marsden-Smedley 2009). Distance sampling and variable circular-plots were used within a replicated space-for-time (SFT) design to compare patterns in avifaunal composition and density across three seasons and two chronosequences in low productivity, blanket moorlands at Lake Pedder (3-54 years post-fire) and medium productivity, eastern moorlands at Lake St Clair (1-44 years post-fire) in the TWWHA. The primary aims of this study were 1) to quantitatively describe the avifauna of buttongrass moorlands; 2) to investigate the short- to long-term effects of different times since fire and other environmental attributes on avifaunal composition and densities; 3) to identify the seral stages that provide suitable habitat for the resident species; and 4) to provide information to help guide fire management and conservation of the buttongrass moorland avifauna.

Methods

Study area

Details of the study design, site descriptions, fire histories, vegetation configurations, and floristics have been provided in Chapters 2-3 but a summary of the results as they pertain to this study is provided below. The study area is largely within the TWWHA, as well as adjacent land allocations, and is composed of a mosaic of buttongrass moorland, scrub, wet sclerophyll woodland and forest, and temperate rainforest. The two study locations include sites on the low productivity (i.e. infertile) quartzitic geologies at Lake Pedder (3-54 years post-fire, $n = 12$; 305-370 m asl) and the moderate productivity doleritic geologies at Lake St Clair (1-44 years post-fire, $n = 14$; 730-795 m asl). Within the study sites, buttongrass moorland comprises the primary vegetation matrix and is mainly composed of Southwestern Buttongrass Moorland and Sedgely Buttongrass Moorland (i.e. blanket moorlands) at Lake Pedder and Eastern Buttongrass Moorland and Subalpine Sedgeland (i.e. eastern moorlands) at Lake St Clair (for descriptions of these vegetation types see Chapter 3). The moorland matrix is interspersed and bordered by typically small perennial watercourses (i.e. first-third order streams; after Strahler 1952) and associated vegetation communities classified as Southwestern Buttongrass Riparian Zones (mean width 12.5 ± 1.5 m, $n = 28$) and Eastern Buttongrass Riparian Zones (mean width 9.2 ± 1.0 m, $n = 51$), respectively. The matrix is also interspersed with wood copses (mean area 0.5 ± 0.1 ha, $n = 14$) classified as Southwestern and Eastern Wood Copses, respectively. The woodland and forest vegetation classes at Lake Pedder are variable, but are most commonly composed of *Eucalyptus nitida* Wet Scrub and Forest, while at Lake St Clair are dominated by *Eucalyptus delegatensis* Woodlands which form relatively distinct edges with the matrix.

Avian survey methods

A considerable portion of the ornithological literature to date has focused on examining survey design and methods for counting birds in order to investigate a wide range of research questions, particularly regarding avian habitat relationships (for reviews see Hewish and Loyn 1989; Schwarz and Seber 1999; Bibby *et al.* 2000; Rosenstock *et al.* 2002; Watson 2004; Buckland 2006). A brief overview of these methods is provided below followed by a detailed description of the methods used for this study.

Avian survey methods range from complete censuses, in which all (or at least most) individuals within a population are counted to determine absolute abundance and/or density, to relative indices, in which individuals and/or cues (e.g. calls) are counted and assumed to represent a constant proportion of the true population size. Implicit in these methods is a general recognition of both the likelihood and ability of observers to count all individuals or cues that are present. A wide range of factors can contribute to a species' (and individual's)

detectability, including call type and frequency, sex, age, size, plumage, behaviour, and life history, as well as environmental variables such as habitat, time of day, season, weather, and background noise (Dawson 1981a; Robbins 1981a, 1981b; Craig and Roberts 2001; Simons *et al.* 2007). In particular, disturbance by fire causes distinct changes in vegetation structure, and therefore may directly or indirectly affect the conspicuousness of birds (Burbidge 2003). The degree to which these factors influence counts must be considered within the context of observer experience and visual and auditory acuity, as well as the probability that an individual is available for detection at the moment that the observer is actively surveying (Diefenbach *et al.* 2007). While some of these factors can be controlled through appropriate study design and implementation, many of these variables are an unavoidable aspect of field work in a natural environment (Thompson 2002).

Reliable abundance estimates are particularly difficult to obtain for cryptic species, such as the three resident species of buttongrass moorlands. A number of alternative methods have been proposed to increase the detectability of such cryptic species. One method is to utilise call playback, which has been used in a number of avian studies (e.g. Marion *et al.* 1981; Lynch 1995), including for the Southern Emu-wren (Pickett 2000). However, regular playback may cause adverse impacts on the populations being studied and may result in positive bias in density estimates since birds may be drawn in towards the observer from a larger and unknown area (Buckland *et al.* 2001). Methods such as territory mapping (Bibby *et al.* 2000), area searches (Loyn 1986), multiple observer approaches (Hutto and Mosconi 1981; Baker and Whelan 1994), and the use of dogs (Buckland *et al.* 2001), may likewise increase the detectability of species by focusing search efforts in relatively small areas, yet their resource-intensive nature, limited scale, and potentially adverse impacts within the TWWHA prohibited their use in this study.

The vast majority of avian studies rely upon some form of index count, in spite of the fact that the limitations of such methods have long been recognised (Rosenstock *et al.* 2002; Diefenbach *et al.* 2003). However, since differences in detectability within a study are often not accounted for and the assumption of constant proportionality is often not met, it is difficult if not impossible to reliably estimate absolute bird abundance and density (Burnham 1981; Norvell *et al.* 2003). Low bias and high precision population estimates are necessary in order to make valid inferences for most research applications (Thompson 2002). Analytical methods that correct such differences in detectability are not new (e.g. Burnham and Anderson 1976; Reynolds *et al.* 1980), but their use is still somewhat limited despite their obvious advantages (Rosenstock *et al.* 2002). Of these methods, one of the most widely used and broadly applicable is distance sampling, which is comprehensively described in the standard references by Buckland *et al.* (2001, 2004) and summarised below.

Distance sampling is an integrated method that includes study design, data collection, and statistical analyses (Buckland *et al.* 2001; Rosenstock *et al.* 2002). It is essentially an extension of quadrat sampling; however, instead of assuming that all birds are detected within a given area, the observer measures the distance (y) from the line or point to each bird that is detected. These distances, or distance categories, are then used to estimate the detection function ($g(y)$), which is based on the premise that the probability of detection decreases with increasing distance from the observer, assuming that all individuals are available for detection (Buckland *et al.* 2001; Diefenbach *et al.* 2007). However, other factors may also affect detectability, such as different survey methods (e.g. line or point), modes of detection (e.g. audio or visual), type of objects (e.g. individuals or clusters), species, locations, times, and survey conditions. Depending upon the particular research objectives, study design, and sample sizes, a specific detection function can be determined for such variables individually or in combination. Using conventional distance sampling (CDS) analysis, the probability of detection based on these distances can then be modelled using robust, semi-parametric methods (i.e. key function + series adjustment) in Distance 5.0 software (Buckland *et al.* 2001; Thomas *et al.* 2006). With multiple covariate distance sampling (MCDS) analysis, additional covariates (e.g. time, season, habitat variables) can also be included in the detection function model through a log link key function and thus can influence the scale of the detection function (Thomas *et al.* 2006; Marques *et al.* 2007). In the absence of sufficient data to model strata- or condition-specific detection functions, the incorporation of covariates may account for variation in detection probability and thus provide a means to increase the reliability and precision of density estimates as well as assess the influence of the covariates themselves (Marques and Buckland 2004; Marques *et al.* 2007). One of the most notable features of these models is that they are ‘pooling robust’, meaning that variables other than distance, including those that have not been measured or modelled *per se*, are incorporated into the detection functions, and thus can affect detectability without biasing density estimates (Burnham *et al.* 1980; Buckland *et al.* 2001; Burnham *et al.* 2004). In other words, detection distance may serve as a proxy for a suite of other factors that may influence the detection process (Alldredge *et al.* 2007). The model(s) with the best fit are then used to correct the site specific encounter rates (i.e. raw counts) for differences in detectability, and thus estimate absolute density and/or abundance. However, a number of key assumptions must be met in order to obtain reliable estimates (Buckland *et al.* 2001). These assumptions include:

- 1) all birds directly on the line or point are detected;
- 2) birds are detected prior to any responsive movement; and
- 3) distances are measured or distance categories are recorded accurately.

Although under some circumstances these assumptions can be relaxed and the method can be robust to violations, proper survey design and field methods are essential to ensure they are met to a reasonable degree (Buckland *et al.* 2001; Norvell *et al.* 2003). Distance sampling has been assessed and validated in a number of studies (e.g. Nelson and Fancy 1999; Cassey and McArdle 1999; Norvell *et al.* 2003) and is considered by many to be a more rigorous, robust, and viable alternative to traditional avian survey methods (e.g. Bibby 2000; Buckland *et al.* 2001; Rosenstock *et al.* 2002; Thompson 2002; Diefenbach *et al.* 2003; Ellingson and Lukacs 2003; Royle *et al.* 2004; Buckland 2006; Buckland *et al.* 2008). However, it should be noted that some researchers have questioned whether or not the underlying assumptions of distance sampling can reasonably be met in most avian ecology studies and that the resulting density estimates may not be as accurate as simple, unadjusted counts (Verner and Ritter 1985; Raphael 1987; Hutto and Young 2003). On the other hand, others propose that relative abundance methods (i.e. indices) often fail to meet the fundamental assumption of constant proportionality (Ellingson and Lukacs 2003; Norvell *et al.* 2003). Thus, the relative advantages and disadvantages of distance sampling are likely to be a source of continued debate.

Since the three resident species of buttongrass moorlands are very cryptic, can often only be detected by call, and vary in their overall detectability, a number of different survey techniques were considered in order to ensure representative and adequate samples in this study. Three common techniques are used to survey bird populations, including line and point transects used in distance sampling, and point counts, which can be used as a type of plot sampling (e.g. variable circular-plots; see below). Their relative advantages and disadvantages have been extensively discussed in the literature (e.g. Ralph and Scott 1981; Ralph *et al.* 1995; Buckland *et al.* 2001). All other things being equal, line transects are considered to be the most efficient technique, and are particularly well suited for species in low densities and detected through a flushing response (Casagrande and Beissinger 1997; Buckland *et al.* 2001; Buckland 2006), such as Striated Fieldwrens (pers. obs.). However, point transects are advantageous when uneven terrain (Dawson 1981b) or other conditions, such as background noise generated from walking through scrubby vegetation, make it difficult to fully concentrate on detecting and recording birds while moving (Reynolds *et al.* 1980). This is particularly relevant in buttongrass moorlands since they are difficult to traverse quietly due to their hummocky and scrubby nature, and when surveying cryptic species, such as the Southern Emu-wren, that are often only detected by their soft and high-pitched calls (see Chapter 1). Point counts have similar advantages to point transects and can be used to determine absolute densities if the radius of detection is either estimated (e.g. based on maximum hearing distance; Bryant 1991) or fixed (Petit *et al.* 1995), and all birds within that area are likely to be detected. Such point counts have typically been used for

surveying Ground Parrots since their relatively loud and distinctive calls can consistently be detected during their dawn and dusk calling-flight sessions and evidence indicates that most individuals will call during these periods (Bryant 1991; McFarland 1991c; Meredith *et al.* 1984).

In consideration of the low expected densities for the three resident species, their differences in detectability, and the large area that needed to be surveyed (782 ha), it was decided to utilise a composite survey design (Casagrande and Beissinger 1997; Buckland *et al.* 2001). This design consisted of line and point transect distance sampling for the avifauna as a whole and variable circular-plot sampling for Ground Parrots in order to estimate bird densities within each site. Ancillary data, including habitat use by birds, were also collected in order to identify bird-habitat relationships both within and across sites of different fire ages (see Chapter 5). A series of pilot studies were conducted in order to establish a standard survey design and methods that were both feasible to implement and would enable the attainment of the overall aims of this study.

Point and line transect distance surveys

An area-proportionate sampling scheme was chosen in order to maximise coverage and hence sample size for the study area, and because area *per se* was one of the potential explanatory factors to be used in subsequent analyses (Atkinson *et al.* 2006). A systematic grid with spacing of 150 m between lines and points was chosen to maximise the detection of the residents, and minimise overlap and potential double counting between points and lines (Buckland *et al.* 2001). This distance was based on a compromise between the expected maximum detection distances of the Southern Emu-wren (~ 50 m) and Striated Fieldwren (~ 100 m), as determined from the literature (see Chapter 1) and the pilot surveys. A random first point was located in each site based on 75 m buffers from the site boundaries at the point of access (considered to be random in relation to theoretical bird distribution) (Buckland *et al.* 2001; Diefenbach *et al.* 2003). The grid was then oriented to maximise coverage probability for all portions of the sites and to strike a balance between maximising line/point replication and survey efficiency within each site (Strindberg *et al.* 2004). Each point was marked with a 1.2 m bamboo stake and survey flagging along the appropriate bearing, and where necessary additional flagging was placed between stakes so that the line could be clearly identified for distance measurements during the course of surveys (Buckland *et al.* 2001). This type of ‘minus sampling’ survey design may result in under-sampling of the edge and uneven coverage probability since the survey plots do not extend beyond the site boundaries and the resulting detection functions reflect both detectability and availability of birds (Strindberg *et al.* 2004; Thomas *et al.* 2006). However, potential biases from these edge effects are considered to be minor and can reasonably be ignored for most

surveys (Buckland *et al.* 2001; Strindberg *et al.* 2004; Buckland 2006). A total of 116 points and 32 transects (totalling 12.60 km) were established at the 12 sites at Lake Pedder and 181 points and 43 transects (totalling 20.28 km) at the 14 sites at Lake St Clair. It is important to note that due to the limited size and configurations of some sites (see Chapter 2) fewer than the ideal number of points/lines per site could be established in order to calculate reliable variance estimates ($\sim \geq 20$; Buckland *et al.* 2001).

All point and line transect surveys were conducted by the author to eliminate the potentially large biases and reduced precision that can result from inter-observer variability (Cunningham *et al.* 1999; Diefenbach *et al.* 2003). Although seasonality *per se* was not of specific interest, surveys were replicated over three seasons to account for within-site variability and increase the sample sizes and hence reliability of estimates of the detection functions and abundances (Link *et al.* 1994; Buckland *et al.* 2001). Surveys were conducted during the summer (2 February - 9 March), winter (24 May - 6 July), and spring (11 October - 27 November) of 2004, such that each site was surveyed three times. These intervals were considered to be of sufficient length to reduce the dependency of stochastic errors between successive surveys (Buckland *et al.* 2001). Surveys were started approximately one hour after sunrise, based on previous research conducted on the resident species (e.g. Maguire and Mulder 2004) and personal observations. They were conducted for up to six hours each day, as determined by the approximate time taken to survey the largest site in its entirety (BCS, 83.45 ha) and personal observations that indicated birds were detectable by both auditory and visual cues throughout this time period. A survey schedule was developed in which the time of day and order of sites were systematically rotated within and across seasons to minimise temporal biases and maximize survey efficiency (Mac Nally 1996a; Campi and Mac Nally 2001). Although it would have been desirable to limit surveying to calm and fine conditions, this was not feasible because of the highly variable weather typical of the study regions (see Chapter 2) and the large area that needed to be surveyed by a single observer. However, surveys were not conducted or were stopped if unfavourable conditions prevailed, such as sustained moderate or stronger winds ($\geq 20 \text{ km h}^{-1}$) and moderate rain, since these conditions can adversely influence bird detectability (Robbins 1981a; Maguire 2005).

Surveys were conducted by using a combination of line and point transects as described by Buckland *et al.* (2001) and the date, sunrise time, start time, and end times were recorded for each survey. Even though distance methodology is considered to be robust to the influences of environmental variables on density estimates (Buckland *et al.* 2001), weather conditions that could affect bird behaviour and detectability were recorded for each survey since at times surveying had to be conducted in sub-optimal conditions. Temperature ($^{\circ}\text{C}$), humidity

(%), and wind speed (km h^{-1}) were recorded at breast height with an electronic weather gauge (Skymate Plus SM-19, Speedtech Instruments; accuracy: wind $\pm 3\%$; rH $\pm 4\%$; $\pm 1^\circ\text{C}$). Temperature and humidity were recorded at the start and end of each survey. Prevailing wind speed was classified as calm (0 km h^{-1}), light ($< 19 \text{ km h}^{-1}$), moderate ($20\text{--}29 \text{ km h}^{-1}$), or fresh ($30\text{--}39 \text{ km h}^{-1}$) (after BOM 2004), and the prevailing direction was noted. Cloud cover was estimated by eye and classified as clear (0%), mostly sunny (25%), partly cloudy (50%), mostly cloudy (75%), or overcast (100%). Precipitation was classified as none, fog, light, moderate, or heavy rain, or snow.

Each point was surveyed for four minutes, which is congruous with the standards recommended for point counts (Ralph *et al.* 1995) as well as the amount of time it typically takes to traverse each 150 m of transect. Counts did not start until approximately one minute after the point was reached to allow for normal bird activity to resume (Pyke and Recher 1985). After the point count, the 150 m of line transect up to the next point was traversed at a slow and consistent speed, and as quietly as possible. Periodic stops were made to listen for auditory cues and to thoroughly scan the area for birds. Line transects were traversed at a mean rate of travel of 1.86 km h^{-1} , including the occasional stops, which is within the range of what is considered to be a reasonable survey speed in open terrain (Baker 1997; Bibby *et al.* 2000). Binoculars (Gerber Montana 8 x 42) were used to aid in identification, improve detection distance, and to increase the chances of visual observations, particularly for the cryptic resident species. While the surveys were being conducted, search effort was focused on and near the point/line to ensure that the probability of detection at the point/line was unity (i.e. $g(0) = 1$; Buckland *et al.* 2001). During the line survey or after the point was completed, and if necessary, the immediate area ($\sim < 25 \text{ m}$) around the point/line was searched for a brief period (up to a few minutes) to confirm species identification, group size, or other data of any birds that were detected (particularly by call) during the survey period. However, if there was reason to believe that the bird(s) moved in response to these searches, an exact distance was not recorded even if there was subsequent visual confirmation. Only birds using habitat within the site as defined were recorded. Therefore, birds flying through the site that did not appear to be feeding or otherwise using the site as habitat were not included. This included aerial feeders, such as White-throated Needletails, if they were flying high above the ground and could not be directly associated with the site itself. In addition, since mature trees in adjacent woodland edges and large wood copses are not typically affected by the moorland surface fires (Marsden-Smedley *et al.* 1999; Pyrke and Marsden-Smedley 2005), birds observed in the primary canopy were not recorded. Conspecifics that were observed in close proximity to each other ($\sim < 10 \text{ m}$ apart) and exhibited behaviours consistent with those of a pair bond or family group were recorded as a cluster. For each individual identified by call the following were recorded: time, species,

cluster size, location (left or right of point), call type (for residents, see Chapter 1), and distance (within or beyond 25 m). In consideration of the limitations of binomial models it would have been desirable to group distances into more than two categories; however, the high-pitched and ventriloquistic calls of some of the species (e.g. Striated Fieldwren; see Chapter 1) and results from the pilot surveys and other studies (e.g. Jarvinen and Vaisanen 1975; Bibby *et al.* 1985; Diefenbach *et al.* 2003; Alldredge *et al.* 2006) indicated that this would have likely resulted in highly variable and unacceptable error rates, thus violating one of the underlying assumptions of distance sampling (Buckland 1987; Alldredge *et al.* 2006). For visual observations, the exact radial distance (m) between the point and bird or exact perpendicular distance (m) between the line and bird was measured with a laser rangefinder (Leica LRF 800 Rangemaster; accuracy ± 1 m, range 10-800 m) or paced if < 10 m (for methods refer to Buckland *et al.* 2001). In most cases, the rangefinder was aimed at a prominent object (e.g. tussock or shrub) nearest to the bird. If a bird was identified ahead or behind of the observer, then the nearest prominent shrub or other landmark was noted and the measurement was taken from the position on the line perpendicular to the observation. In the relatively few cases in which the birds were a long distance from the observer, the angle and bird-observer distance were recorded and were later used to calculate perpendicular distances. Visual observations of aerial feeding species, such as Welcome Swallows and Tree Martins, were only categorised as being within or beyond 25 m due to the difficulty in accurately measuring their exact distance when initially observed.

In addition to the above information, sex (for dichromatic species), age (when discernible), behaviour (flushed, feeding, flying, perching), and habitat type (see Chapter 5) were also recorded for visual observations. If a bird was flushed while traversing the line, the bird-observer distance and flight length were also measured or estimated by eye. Care was taken to avoid double counting of individual birds (particularly at the same point or along the same line) by noting their relative location (e.g. left or right of transect) and maintaining a mental map of their locations. However, distance sampling is considered to be robust to double counting between separate points and lines, and typically neither contributes much bias nor violates any underlying assumptions of the method (Buckland *et al.* 2001; Buckland 2006). The above process was repeated until all point and line transects had been surveyed once within the site.

Variable circular-plot surveys

A variable circular-plot (VCP) survey method (i.e. point count), as outlined by Reynolds *et al.* (1980) and modified by Bryant (1991), was used to survey sites for Ground Parrots during each of the three seasons since they could not be reliably detected during the course of the daytime distance surveys (see Chapter 1). Surveys were conducted during their dusk

calling-flight sessions since they are considered to be most detectable during this period and all birds at a site are thought to call during each session (Bevege 1968; Meredith *et al.* 1984; Jordan 1989; McFarland 1991b; Burbidge *et al.* 2007). This method has proven to be both reliable and effective due to the predictability of these calling sessions and the long distance over which calls can typically be heard. Furthermore, results from a number of studies utilising this and similar methods have been corroborated by a range of different techniques including repeat surveys conducted over several consecutive days, thorough area searches, and opportunistic observations (Meredith *et al.* 1984; Bryant 1991; McFarland 1991c). An additional benefit is that Bryant (1991) utilised this method for surveying Ground Parrots in Tasmania during 1989-90, thus results from this study should be directly comparable and usable by fire and conservation managers. All surveys were conducted at the first point (i.e. point #1) at each site, which by design was adjacent to site and habitat boundaries. Although this limited the proportion of sites from which calls could be heard, it made it easier to confirm whether a given call was 'in' or 'out' of the site. Surveys were started at sunset, and weather conditions at the start of the survey were recorded as above. Based on the results of previous research and a pilot study, surveys were conducted for 30-60 minutes, which allowed for enough time to establish presence-absence and to count all calls issued during a given session (Bryant 1991; McFarland 1991b; Burbidge *et al.* 2007). The survey area was scanned in all directions while standing quietly at the point (Bryant 1991; Burbidge *et al.* 2007). The time and relative location of all calls and flight observations were noted on a datasheet that included a basic map of the site. While it would have been desirable to estimate the distance from the point to each observation to use in distance sampling, as described above, results from a series of pilot studies indicated that distances to aural cues could not be estimated with any acceptable level of accuracy. Volunteer observers were used for repeat surveys at sites that had already been surveyed by the author and where Ground Parrots were thought to be absent or unlikely to occur. All volunteers were trained by accompanying the author on surveys where Ground Parrots were known to be present to ensure that they could both hear and identify the calls, and were familiar with the site boundaries, methodology, and overall objectives. In consideration of their limited use and training, it is unlikely that they contributed significant variability to the results (Kepler and Scott 1981).

Survey conditions

Weather conditions during the surveys were typical for the study area (see Chapter 2); temperature and rainfall data for 2004 compared to historical climatic data (1961-1990) indicated that there were no major anomalies at either location (BOM 2007; I. Barnes-Keoghan pers. comm. 2007). The mean temperature was 10.8°C (range 2-23°C) during the daytime surveys and 9.4°C (range 1-20°C) at dusk, while mean relative humidity was 74.7%

(range 40-100%) during the daytime and 80.8% (range 49-100 %) at dusk. During both daytime and dusk surveys cloud cover and wind speeds were highly variable. Overall, survey conditions were typically partly cloudy to overcast (65%; $n = 156$) with calm to light winds (85%) and no precipitation (69%). The potential influence of survey-specific weather conditions are incorporated into the density estimates and analyses, as described below.

Distance density estimates

Point and line transect data were analysed using Program Distance 5.0 (Thomas *et al.* 2010) following the methods detailed by Buckland *et al.* (2001, 2004) and Thomas *et al.* (2006, 2010). Four separate sets of analyses were conducted, including for exact points, exact lines, binomial points, and binomial lines. Although sample sizes were considerably larger for the binomial datasets and hence generally more precise due to the inclusion of both auditory and visual observations, for these datasets fewer than 50% of observations were within the first distance bin, thus limiting the possibility of obtaining robust estimates (Buckland 1987). Based on an assessment of preliminary analyses, the binomial density estimates are not presented or discussed herein. However, since the binomial data contained fewer absences overall, they were used to determine species occurrence in conjunction with the exact data, such that a species was considered to be present if recorded at a site during any one of the seasonal surveys and absent if not recorded during any survey. In the case of the resident species, some off-survey observations indicated that they were present at a few sites in which they were not detected during the course of the actual surveys. Thus, it was apparent that they occasionally used more sites than indicated by the presence-absence data (i.e. detection-nondetection; MacKenzie 2005). Nevertheless, since such opportunistic observations indicate ‘use’ but not necessarily ‘occupancy’, they were not included in subsequent analyses (MacKenzie 2005).

Exact data were pooled from the repeat seasonal visits to a single line or point since they were originally intended to provide an overall estimate of density across the year. Thus, survey effort was multiplied by three for points (total number of visits, $K = 837$) and lines (total length, $L = 96,810$ m), accounting for points that fell near site boundaries or for lines that passed through non-habitat (i.e. woodland). Since the points along transects were spaced equidistant to lines (150 m), the individual points were taken as the sampling unit (Buckland *et al.* 2001; Thomas *et al.* 2002). Distance analyses were conducted separately for the resident Southern Emu-wren and Striated Fieldwren since they were the primary focus of the distance surveys and both exploratory analyses and personal observations indicated that their detectabilities differed.

A number of non-resident aerial feeding and hunting species were excluded from the distance analyses since they were only recorded as binomial observations. Furthermore, the survey methods were subsequently determined to be poorly suited to obtaining reliable density estimates (e.g. due to their high mobility) and their associations with the sites themselves proved to be tenuous (Buckland *et al.* 2008). These included unspecified raptor species, Tree Martin (*Hirundo nigricans*), Welcome Swallow (*Hirundo neoxena*), and White-throated Needletail (*Hirundapus caudacutus*). Similar to these latter species, nine others were present in less than 5% of point/line transect surveys across the year ($n = 78$), thus the limited number and opportunistic nature of these observations did not warrant their inclusion (McCune *et al.* 2002; see Results). The remaining 10 species (see Table 4), comprising the non-resident group, were analysed together due to the small sample sizes and the similarity in detectability for most of these species. For example, they are generally conspicuous (e.g. maximum detection distances > 85 m), noisy, and associated with early successional or edge habitats (Higgins 1999; Higgins *et al.* 2001; Higgins and Peter 2002; Watts 2002; Higgins *et al.* 2006; pers. obs.). Alldredge *et al.* (2007) used a similar method and concluded that such a parsimonious multi-species approach may produce more precise density estimates without a substantial increase in bias.

The following methods were used to build, test, and select detection function models for each set of exact point and line transect analyses. Histograms of the number of observations in relation to distance (y m) for the Southern Emu-wren, Striated Fieldwren, and non-resident species were inspected and data were right-truncated (at w m) for each analysis in order to better model the detection functions and since outlier observations (i.e. $> w$ m) typically do not contribute much information towards their estimation (Buckland *et al.* 2001). All detections were treated as clusters and cluster sizes (s) were estimated at the global level due to small sample sizes. Size-biased regression (i.e. $\ln(s)$ vs. $\hat{g}(y)$) was used to correct for the possibilities of size-biased detection and the underestimation of the size of detected clusters (Buckland *et al.* 2001). The encounter rates of clusters ($n K^{-1}$ or $n L^{-1}$) were estimated at the stratum level while the detection functions were estimated at the global level (i.e. including all sites at both locations) due to small sample sizes. A range of plausible detection function models was assessed using the uniform, half-normal, and hazard-rate key functions with cosine, simple polynomial, and hermite polynomial series expansions. The shape of the fitted functions were assessed with either no constraints or with strictly or weakly monotonically non-increasing constraints. All possible adjustment terms (≤ 5) were assessed and selected using Akaike's Information Criterion (AIC, see below) (Buckland *et al.* 2001; Thomas *et al.* 2006). Using the MCDS engine, a range of covariates at the stratum and observation layers were assessed that were considered *a priori* as most likely to affect detectability in respect to, but independent of distance, including: location, fire age

(continuous and categorical), cover index, season, and time (Thomas *et al.* 2006; Marques *et al.* 2007). Contrary to the Ground Parrot analysis, wind speed and precipitation were not included as variables since detections were not solely dependent upon auditory cues. For the MCDS analyses, detection functions were estimated at the global and stratum levels and only the half-normal and hazard-rate key functions with no constraints were available for testing (Thomas *et al.* 2006). Model fit was assessed using the Kolmogorov-Smirnov goodness-of-fit test (GOF) and q-q plots of the fitted cumulative distribution function (cdf) against the empirical distribution function (edf), and by visual inspection of the detection probability plots and probability density plots (in the case of point transects) (Burnham *et al.* 2004; Thomas *et al.* 2006). Although sample sizes were small, the relative fits of candidate models were assessed using AIC instead of AICc since the former does not assume the data are normally distributed (Buckland *et al.* 2001). The weight of evidence for each model was assessed using the differences between AIC values ($\Delta AIC = AIC - AIC_{\min}$) and Akaike weights (Burnham and Anderson 2002). In the case of the MCDS analyses the failure to include model covariates may result in higher precision at the expense of substantial bias of habitat-specific density estimates, since AIC attempts to strike a balance between decreasing bias by adding more parameters and increasing precision by decreasing the number of parameters (i.e. the principle of parsimony) (Buckland *et al.* 2001; Burnham and Anderson 2002; Marques and Buckland 2004; Marques *et al.* 2007). Therefore, histograms, GOF tests, and other associated evidence were also evaluated for models with comparable AIC values to determine whether another model was more plausible (Buckland 2006).

The best model for each set of analyses was used to estimate the probability (\hat{P}_a) of the number of clusters detected within the surveyed area (a), effectively dealing with the issue of constant proportionality, and thus allowing for estimates of population densities based on site-specific encounter rates (Buckland *et al.* 2001; Norvell *et al.* 2003). Incorporating the expected cluster sizes for the selected models, the density estimates (\hat{D}) were calculated at the stratum level. In the case of the non-resident species, \hat{P}_a and the standard error (SE) of \hat{P}_a were calculated for the shared-detection functions and then used as multipliers with a uniform key function and no adjustment terms to calculate the density of each species for each site for use in the multivariate analyses, as Distance is currently not capable of handling multiple levels of stratification (Thomas *et al.* 2006).

Preliminary analyses indicated that the estimated detection functions and encounter rates differed between the point and line transect surveys for the Southern Emu-wren, Striated Fieldwren, and non-resident group. Furthermore, they were not consistently detected from both the point and line transects for all sites. However, for all sites where they were

recorded from both transects, the methods produced species- and group-specific density estimates with 95% lognormal confidence intervals that overlapped. In part due to the small sample sizes, it was difficult to determine from the detection functions and ancillary data which of the methods best met their respective assumptions and thus provided the least biased estimates of density. Due to these issues, the estimated species- and group-specific densities derived from the point and line methods were averaged for each site (mean birds ha^{-1}) (Buckland *et al.* 2001). The variance of density estimates was approximated using the delta method and included variance from the encounter rate, detection probability, and cluster size estimation (Buckland *et al.* 2001). Estimates of encounter rate variance were based on a Poisson distribution and each point and line was treated as a replicate (Buckland *et al.* 2001; Thomas *et al.* 2002). Since a number of the sites were small and/or linearly-shaped and only had one or a few line transects and few points (see Chapter 2), it was not possible to obtain variance estimates for such sites. Furthermore, since count data are often overdispersed (e.g. patchily distributed) due to a lack of complete independence and other sources of heterogeneity, an overdispersion parameter (b ; or variance inflation factor) was calculated for each set of analyses following the method described by Buckland *et al.* (2001) in order to provide more reliable variance estimates for all of the sites (Lebreton *et al.* 1992; Burnham and Anderson 2002; Franklin *et al.* 2002). Estimates of detection probability and cluster size variances were calculated in Distance using the methods of Buckland *et al.* (2001). Estimates of variance for the final combined point and line density estimates for each species and group by site were derived following the method of Steel and Torrie (1980).

VCP density estimates

The number of individuals present at each site was estimated by plotting the number of spatial and temporal clusters of calls recorded on the datasheets to calculate the occurrence and relative densities of Ground Parrots. They were considered to be present if recorded at a site during any one of the seasonal surveys and absent if not recorded during any survey. Under some circumstances it was difficult to determine whether calls were being issued by the same or different birds, particularly since they fly around during these sessions (Burbidge *et al.* 2007); therefore, simultaneous (Chan and Mudie 2004) and responsive calls (Jordan 1987b) were relied upon to distinguish individuals. Standard hearing distances as estimated by Bryant (1991) using call playback were used to determine the maximum call detection distance in a range of weather conditions known to affect detectability (Table 1). This approach has been utilised by other avian studies (e.g. Emlen and DeJong 1981) and the distances reported, although crude, are consistent with personal observations and results from previous studies of the Ground Parrot (Bevege 1968; Meredith and Isles 1980; Meredith *et al.* 1984; Watkins 1985; K. Chan pers. comm. 2005; D. McFarland pers. comm.

2005; Burbidge *et al.* 2007; Gibson *et al.* 2007). The weather data for each survey were used to calculate the radius of each hearing circle, which was then plotted in ArcView GIS (ESRI 2002) and used to determine the estimated area surveyed within each site and for each survey. The number of individuals for each survey was divided by these area estimates in order to calculate an index of relative mean densities of Ground Parrots (birds ha⁻¹). For each site, these estimates were then averaged across seasons (including zero values) since repeat surveys were originally intended to increase sample size and accuracy, and not to provide insights into seasonality *per se*. As above, the variances of density estimates for the sites were calculated from data pooled across seasons and based on a Poisson distribution with $b = 1$, since preliminary analyses did not indicate the data were notably overdispersed.

Table 1. Standard hearing distances (m) for Ground Parrot calls in various weather conditions (after Bryant 1991).

Wind speed (km h ⁻¹)	No rain	Light rain	Medium rain
Calm (0)	400	350	300
Light (< 19)	350	300	250
Moderate-Fresh (29-39)	300	250	200

Univariate analyses of habitat associations

Avifaunal densities were analysed in relation to a range of site-specific environmental variables as reported in Chapters 2-3. However, due to limited sample sizes and the stratified study design, analyses were limited to a subset of variables identified *a priori* as most likely to influence bird-habitat associations based on previous research, ecological theory, and personal observations. These continuous variables included site fire age (years post-fire), cover index, area (ha), edge contrast index (%), and riparian area (%). Other fire regime parameters (e.g. intensity, frequency, season) were not included since verifiable data were not available for all sites (see Chapter 2). Although categorical fire age classes are used in some analyses in this thesis, fire age was treated as a continuous variable here due to the limited degrees of freedom, and since as a continuous variable it is both more sensitive to detecting significant effects and the results are easier to interpret (Quinn and Keough 2002). Since preliminary analyses indicated that the cover index was an important explanatory variable, one site (GIT) was excluded from the analyses reported herein since no cover data were available and the value could not be reasonably extrapolated from other sites (M. Driessen unpublished data). However, exploratory analyses indicated that its exclusion did not have a prominent effect on the overall results. Fire age and cover were highly correlated with each other (see Chapter 3); nevertheless, they were both included in the analyses since previous research on the Ground Parrot and other ground-dwelling faunal species indicated that the resident species may be more responsive to changes in cover than fire age *per se* (e.g. Fox 1982; Meredith *et al.* 1984; Baker and Whelan 1994; Monamy and Fox 2000).

Riparian area was not used in the Ground Parrot analyses since survey data and field observations did not indicate that they rely upon riparian habitats and the majority of their diet is considered to be comprised of graminoid species that are more dominant throughout the moorland matrix (Bryant 1994; see Chapters 3 and 5). However, due to the patchy distribution and relatively high mobility of Ground Parrots, analyses for the Ground Parrot also included 400 m radius circular plots (based on maximum hearing distance; ~ 50 ha) around each survey point and thus extending beyond the site boundaries (i.e. landscape scale). The percentage of suitable habitat was calculated from the WHAveg GIS layer (TVMP 2004) and included vegetation communities that consisted of suitable structural and floristic attributes (e.g. sedgeland, heathland, and graminoid heathland communities; see Chapter 3), as identified by Bryant (1991, 1994). This approach is commonly used in avian research to calculate landscape-scale metrics, particularly for studies investigating factors that influence species occurrence (e.g. Meyer and Miller 2002; Grand *et al.* 2004).

The influence of the above environmental variables on the estimated densities of the Southern Emu-wren, Striated Fieldwren, Ground Parrot, and non-resident group was tested using linear regression since examination of scatterplots and preliminary analyses did not indicate the presence of nonlinear relationships (Quinn and Keough 2002). Consistent with the design of this study, the survey data were pooled across seasons and analysed separately at Lake Pedder and Lake St Clair (see Chapter 2). This set of analyses was restricted to univariate tests since the limited dataset and low number of replicates for the two chronosequences resulted in insufficient degrees of freedom to utilise multiple regression and to explore the potential influence of all possible main and interaction effects. While the Southern Emu-wren, Striated Fieldwren, and non-resident group occurred at most sites, the Ground Parrot dataset contained a large number of zeros and was positively skewed since over the course of the three survey seasons it occurred in low numbers (≤ 6 per survey) and was only present at 7 out of 12 sites at Lake Pedder and 5 out of 14 sites at Lake St Clair. Such multimodal distributions may reflect both the probability of occurrence and the level of abundance in relation to species and environmental heterogeneity (Welsh *et al.* 1996; Quinn and Keough 2002; Fletcher *et al.* 2005). Under such circumstances, Fletcher *et al.* (2005) recommended creating two datasets; the first includes presence-absence data and is analysed with logistic regression, and the second includes abundance data and is analysed with ordinary linear regression, which they also concluded is more informative than using a single-model approach. A similar approach is used herein for the Ground Parrot, but since the ratio of replicates, where present, to explanatory variables was too low to obtain reliable results, it was necessary to include the absences in the density dataset as well. Accordingly, the logistic and ordinary models for the Ground Parrot are not independent, but the results

from the ordinary models should indicate which variables may be influencing Ground Parrot density responses in lieu of or in addition to those identified in the logistic models.

All analyses were conducted in SPSS 14.0 (SPSS Inc. 2005). Both dependent and explanatory variables were examined for normality, and densities for the Southern Emu-wren, Striated Fieldwren, and non-resident group were subsequently square-root transformed ($\sqrt{}$) and densities for the Ground Parrot \log_{10} -transformed by adding a small constant to all values ($y + 0.001$) since zeros were included in the dataset. Preliminary analyses for the Ground Parrot indicated that the overall results were consistent regardless of the constant used. Adequacy of the ordinary regression models regarding the assumptions of normality and homogeneity of variance was checked by inspection of residual plots, which indicated no obvious issues with the models. High influence and high leverage outliers were identified by examining Cook's D statistic versus centred leverage values. In the case of the logistic models, residuals were also examined to identify outlier sites and the Hosmer-Lemeshow test was used to assess goodness-of-fit since all of the predictor variables were continuous (Quinn and Keough 2002). In the few cases where it was warranted, analyses were also conducted with outlier sites excluded.

Multivariate analyses of habitat associations

A notable issue that arises when using a set of univariate (or multivariate) tests is trying to identify the most important explanatory variable(s) that may be inferred to have a causal influence on the response variable (Mac Nally 2000; Quinn and Keough 2002). When the primary interest is to help explain observed patterns as opposed to develop a predictive model, it is necessary to concurrently consider all possible models (Mac Nally 2000). This can be accomplished by using hierarchical partitioning (Chevan and Sutherland 1991; Mac Nally 1996b). Hierarchical partitioning is used to examine the dataset within a multivariate framework by using all possible model combinations from the hierarchy of regression models (i.e. simplest to most complex) and partitions the proportions of both independent and joint variance for each potential explanatory variable based on an appropriate goodness-of-fit measure (Chevan and Sutherland 1991; Mac Nally 1996b, 2000). This method enables the identification of the variable(s) with the greatest relative average independent influence on the dependent variable (positive or negative), as well as their relative joint contributions (i.e. correlations) with the other independent variables (Chevan and Sutherland 1991; Mac Nally 1996b; Quinn and Keough 2002). Although multiple regression could not be used for these datasets, the hierarchical partitioning approach holds advantages over other commonly used multivariate model selection techniques (e.g. sequential) since it is less likely to produce inconsistent and spurious results due to statistical anomalies, high Type I error rates, and issues associated with collinearity (Mac Nally 1996b, 2000; Burnham and Anderson

2002). Furthermore, hierarchical partitioning can be used to make direct comparisons between results of different statistical methods (e.g. ordinary and logistic models), but it is important to note that it does not test null hypotheses or attribute causality *per se* (Chevan and Sutherland 1991; Mac Nally 1996b; Quinn and Keough 2002). Hierarchical partitioning was conducted in R 2.5.0 (R Development Core Team 2006) with the hier.part package (Walsh and Mac Nally 2007) using a Gaussian distribution and R^2 for the density dataset and binomial distribution and log-likelihood χ^2 goodness-of-fit statistic for the Ground Parrot presence dataset.

The influence of the environmental variables (i.e. site fire age, cover, total area, edge contrast index, riparian area) on observed patterns in species composition and densities at the community level was explored using multivariate techniques available in Primer 5.2.2 (Primer-E Ltd. 2001). The similarities of mean densities contributed by species between sites were calculated using the Bray-Curtis measure as it ignores joint absences, while the similarities of each environmental variable between sites were calculated using Euclidean distances (Clarke and Gorley 2001; Quinn and Keough 2002; McCune *et al.* 2002). The avifaunal data were square-root transformed in order to reduce the chances that the resident species would obscure contributions from the less abundant non-resident species (Clarke and Warwick 1994; McCune *et al.* 2002). The RELATE procedure (999 random permutations) was used to test for significant relationships between the similarity matrix for the avifauna and those for each environmental variable with the Spearman rank correlation coefficient (r_s), which is a Mantel-type test where r_s ranges from 0 for no relationship to 1 for a perfect correlation (Clarke and Gorley 2001). The BIO-ENV procedure was then used to determine which environmental variables best matched the patterns in avifaunal composition and densities for each location by identifying the subset of variables that maximised the rank correlation value (Clarke and Warwick 1994). The similarity matrix for the full set of environmental variables was calculated using normalised Euclidean distances since they were measured on different scales (Clarke and Gorley 2001). Finally, the SIMPER procedure (similarity percentages) was conducted to examine the mean one-way similarities in square-root transformed density within Lake Pedder and Lake St Clair and thus determine which species were contributing the most to the avifaunal patterns observed at each location (Clarke and Gorley 2001).

Results

Avifaunal composition

Total survey effort for this study consisted of 78 surveys each for the point transect distance surveys, line transect distance surveys, and VCP surveys, and were conducted at 26 sites across three seasons in 2004. Distance surveys were conducted at a total of 867 points and 219 line transects (totalling 97 km), and along with the VCP surveys accounted for approximately 170 hours of on-survey observation time. A total of only 25 species was recorded, in addition to unspecified raptor species. The 13 resident and non-resident species that were included in subsequent analyses are listed in Table 4. The migratory Tree Martin (*Hirundo nigricans*) and Welcome Swallow (*Hirundo neoxena*) were frequently observed foraging over the sites during spring and summer, but as discussed above were excluded from analyses. The only species that were recorded in this study but not reported in the literature as occurring in moorlands and related habitats were the Grey Shrike-thrush (*Colluricincla harmonica*) and Black-headed Honeyeater (*Melithreptus affinis*). However, these were among ten species that were detected in < 5% of point/line transect surveys across the year ($n = 78$) and will not be discussed any further herein, including: Beautiful Firetail (*Stagonopleura bella*), Black-faced Cuckoo-shrike (*Coracina novaehollandiae*), Grey Fantail (*Rhipidura fuliginosa*), Olive Whistler (*Pachycephala olivacea*), Pink Robin (*Petroica rodinogaster*), Strong-billed Honeyeater (*Melithreptus validirostris*), White-throated Needletail (*Hirundapus caudacutus*), and Yellow Wattlebird (*Anthochaera paradoxa*).

Model selection and detection functions

Data on the 12 species included in the distance analyses (Table 4) were limited to observations of singles or clusters of birds that were confirmed by visual cues and measured using exact distances from the point and/or line transects ($n = 548$ within w , Table 2). Separate detection functions were modelled for the Southern Emu-wren, Striated Fieldwren, and non-resident group data and are presented in Tables 2-3 and Figures 1-3. The results from the model selection process confirmed that different truncation points (range 37-150 m) and models (i.e. half-normal, hazard-rate, hazard-rate + Hermite polynomial) were required to provide the best fits for the species- and group-specific datasets, and for five out of the six datasets the model with the smallest AIC value was selected, all of which provided good fits to the data (K-S GOF $P > 0.5$). A range of covariates that may have affected detectability were tested and both Δ AIC and Akaike weight values indicated that the inclusion of a number of the covariates had some support from the resident species datasets (Δ AIC < 2; Akaike weights > 1). However, only in the case of the Striated Fieldwren recorded from point transects was a model that included a covariate (i.e. recent burn) selected as the best by

AIC ($\Delta AIC = 0$; Akaike weight = 0.760). Results for the Striated Fieldwren recorded from line transects indicated that the no covariate model provided the best relative fit; however, the density estimates for the recently burnt sites (i.e. MEE, MEW, HAR) appeared to be positively biased since the model did not account for the higher detectability of the Striated Fieldwren in the open, early successional habitat. Therefore, consistent with the point transects the model that included the recent burn covariate was selected for the line transects in order to provide a more plausible detection function and more accurate density estimates, particularly since it similarly had substantial empirical support ($\Delta AIC = 1.25$, Akaike weight = 0.18) and still provided a good overall fit to the data (K-S GOF $D_n = 0.173$, $P = 0.161$).

The estimated probabilities of detecting a cluster of Southern Emu-wrens, Striated Fieldwrens, and non-resident species within the surveyed areas (\hat{P}_a) and the estimated effective detection radii ($\hat{\rho}$) for points and effective strip half-widths ($\hat{\mu}$) for lines as derived from the selected models are presented in Table 3. The probability of detecting clusters varied due to inter-specific differences in detectability and the associated detection functions, truncations widths, and differential responses to the method of detection. While the detection probabilities for the Striated Fieldwren were similar between line and point transects, those for the Southern Emu-wren were approximately three times higher and those for the non-residents were approximately twice as high for line versus point transects. The effective strip half-widths and effective detection radii were similar between lines and points for both the Southern Emu-wren and non-residents, but the detection distance for the Striated Fieldwren was substantially narrower for line versus point transects ($\hat{\mu} = 13.48$ m, $\hat{\rho} = 86.35$ m). Larger values were expected for the point transects *a priori* since the area and hence number of birds that can be detected increases linearly with distance from the point (Buckland 2006). Nevertheless, the magnitude of the difference for the Striated Fieldwren along with the fact that > 90% of observations within 14 m from line transects were classified as flushing responses, while none were classified as flushing from points regardless of distance, which suggested the species exhibited differential responses in relation to a moving versus stationary observer.

Table 2. Model selection of distance detection functions ($g(y)$; pooled across three seasons) for the Southern Emu-wren, Striated Fieldwren, and non-resident species by point ($n = 867$) and line transect ($n = 219$) surveys conducted in buttongrass moorlands at Lake Pedder ($n = 12$) and Lake St Clair ($n = 13$), Tasmania. Covariates tested included: location, fire age (years post-fire), recent burn (≤ 3 or > 3 years at Lake Pedder, ≤ 1 or > 1 year at Lake St Clair; based on fire age, cover, and pers. obs.), cover index, season (summer, winter, spring), and time (hours since sunrise for observations). Model fit based on Kolmogorov-Smirnov goodness-of-fit statistic (K-S GOF; all non-significant) and model selection guided by delta Akaike's Information Criterion ($\Delta AIC = AIC - AIC_{\min}$) and Akaike weights (Burnham and Anderson 2002) based on the full set of candidate (non-)covariate models tested for each set of analyses. Only models with substantial empirical support (i.e. $\Delta AIC < 2$; Burnham and Anderson 2002) are reported and selected models are in **bold**.

Species	Method	Truncation (w m)	No. of clusters w/in w	Expected cluster size (s)	Model (Key + adjustment, order)	Covariates	No. of parameters	K-S GOF		ΔAIC	Akaike weights
								D_n	P		
Southern Emu-wren	Point	61	27	1.7	Hazard-rate	None	2	0.080	0.996	0.00	0.572
		61	27	1.6	Hazard-rate	Time	3	0.107	0.917	1.15	0.181
		61	27	1.8	Hazard-rate	Cover	3	0.091	0.978	1.81	0.094
	Line	42	20	1.7	Half-normal	None	1	0.149	0.768	0.00	0.377
		42	20	1.6	Half-normal	Time	2	0.130	0.886	0.51	0.226
		42	20	1.5	Half-normal	Season	3	0.120	0.935	0.69	0.189
		42	20	1.7	Half-normal	Location	2	0.192	0.450	1.58	0.078
		42	20	1.7	Half-normal	Fire age	2	0.162	0.668	1.66	0.072
		42	20	1.7	Half-normal	Cover	2	0.132	0.878	1.87	0.058
Striated Fieldwren	Point	150	46	1.3	Hazard-rate	Recent burn	3	0.113	0.596	0.00	0.760
		150	46	1.3	Hazard-rate	Location	3	0.081	0.926	1.89	0.115
	Line	37	42	1.1	Hazard-rate + Hermite, 4	None	3	0.095	0.845	0.00	0.627
		37	42	1.1	Hazard-rate + Hermite, 2	Recent burn	4	0.173	0.161	1.25	0.180
		37	42	1.1	Hazard-rate + Hermite, 2	Fire age	4	0.118	0.606	1.78	0.106
Non-residents	Point	150	78	1.7	Hazard-rate	None	2	0.049	0.992	0.00	0.628
	Line	150	51	1.5	Hazard-rate	None	2	0.080	0.904	0.00	0.630

Table 3. Estimated probability of clusters being detected within the surveyed area (\hat{P}_a), their effective detection radii (EDR, Points) and effective strip half-widths (ESW, Lines), and associated lognormal 95% confidence intervals (CI) from the selected models (see Table 2) by species and method. The EDR/ESW indicate the estimated distance from the point/line ($\hat{\rho} / \hat{\mu}$) at which as many individuals or clusters are missed within $\hat{\rho} / \hat{\mu}$ as detected beyond $\hat{\rho} / \hat{\mu}$ (Buckland *et al.* 2001).

Species	Method	Detection probability		EDR/ESW	
		\hat{P}_a	95% CI	$\hat{\rho} / \hat{\mu}$ (m)	95% CI
Southern Emu-wren	Point	0.17	0.06 - 0.48	25.35	14.88 - 43.18
	Line	0.60	0.41 - 0.88	25.23	17.15 - 37.12
Striated Fieldwren	Point	0.33	0.25 - 0.44	86.35	75.20 - 99.15
	Line	0.36	0.26 - 0.52	13.48	9.50 - 19.12
Non-residents	Point	0.23	0.15 - 0.36	71.91	57.06 - 90.62
	Line	0.52	0.42 - 0.64	77.42	62.72 - 95.57

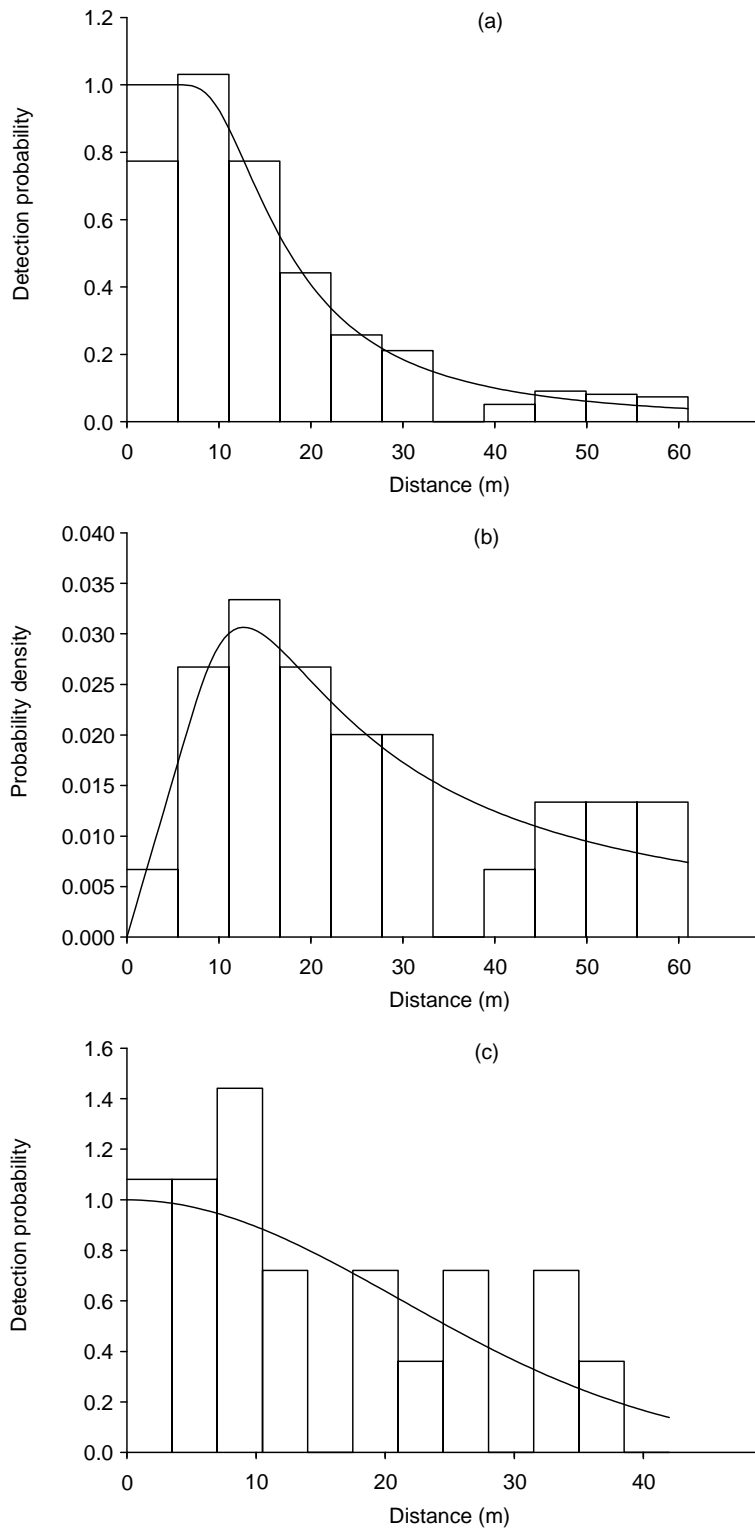


Fig. 1. Southern Emu-wren: (a) The estimated detection probability function and corresponding (b) estimated probability density function from point transect surveys, and (c) estimated detection probability function from line transect surveys based on the selected models (Table 1) for Southern Emu-wren observations in buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. The models are fitted to scaled histograms of observations as a function of distance (m) from the survey points/lines.

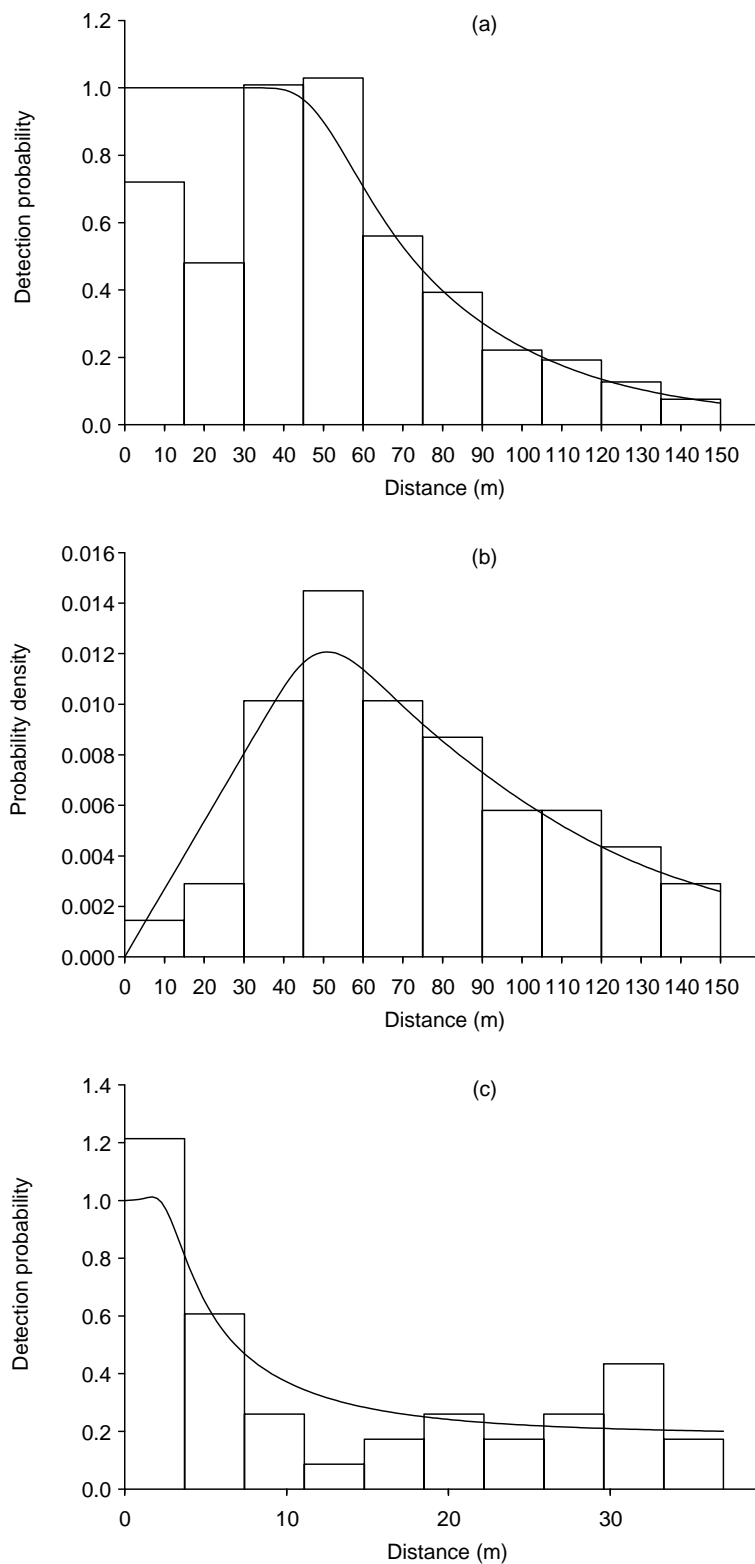


Fig. 2. Striated Fieldwren: (a) The estimated detection probability function and corresponding (b) estimated probability density function from point transect surveys, and (c) estimated detection probability function from line transect surveys based on the selected models (Table 1) and averaged over the recent burn covariate factors for Striated Fieldwren observations in buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. The models are fitted to scaled histograms of observations as a function of distance (m) from the survey points/lines.

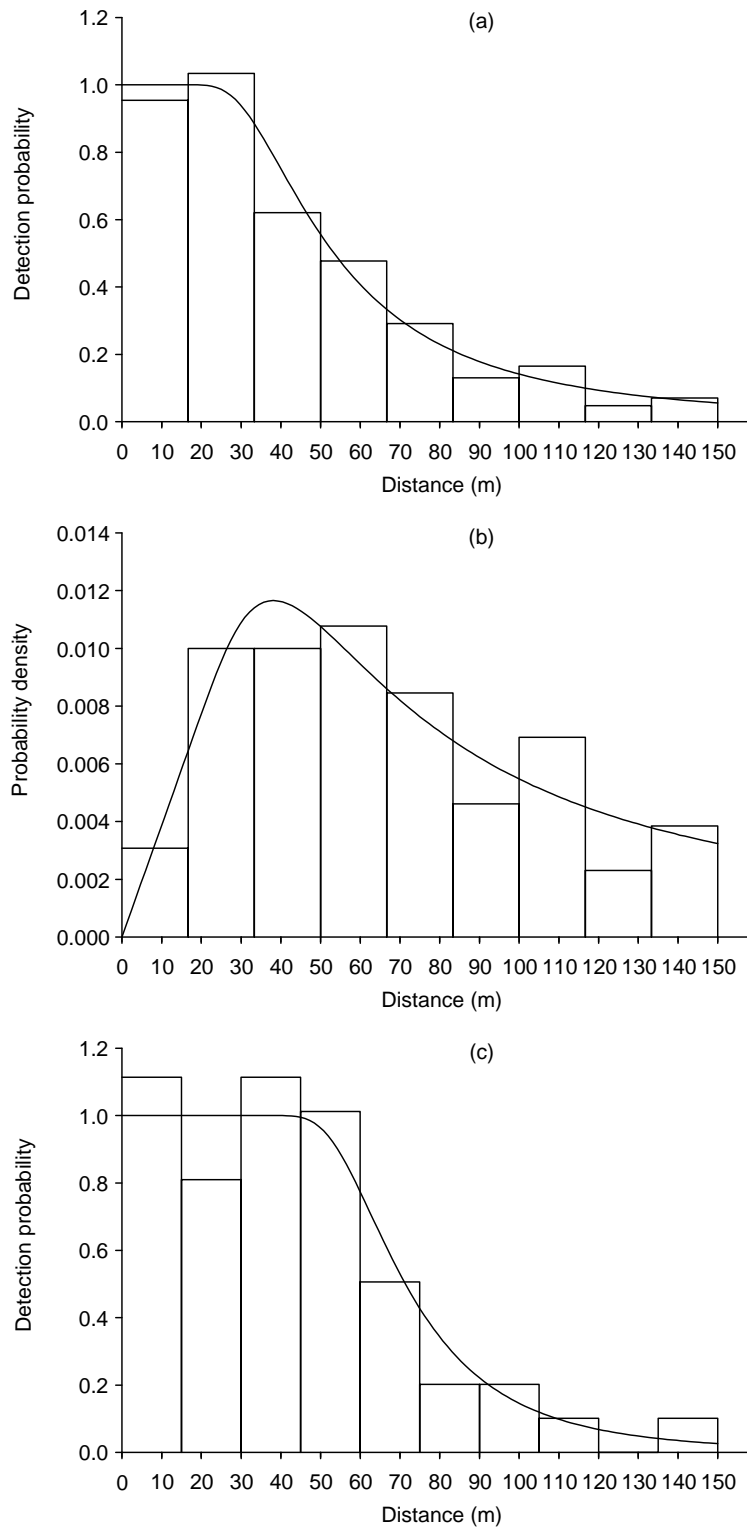


Fig. 3. Non-resident species: (a) The estimated detection probability function and corresponding (b) estimated probability density function from point transect surveys, and (c) estimated detection probability function from line transect surveys based on the selected models (Table 1) for non-resident species observations in buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. The models are fitted to scaled histograms of observations as a function of distance (m) from the survey points/lines.

Species occurrence and density estimates

A summary of species occurrence and estimates of avian population density by species and location are presented in Table 4 and general trends in relation to fire age are presented in Figure 4. The Striated Fieldwren occurred at all but one site (96%; $n = 25$), while the Southern Emu-wren was also widely distributed (88%). The Ground Parrot was patchily distributed (48%), while the remaining species, except for the Tasmanian Thornbill, were present in $\leq 50\%$ of the sites. However, on-survey auditory observations as well as occasional off-survey observations indicated that both use and occurrence at sites, particularly for the resident species, may be somewhat higher than indicated by these figures. The Southern Emu-wren had the highest estimated densities at both Lake Pedder and Lake St Clair (1.83 and 1.64 birds 10 ha^{-1} , respectively), followed by the Striated Fieldwren (1.08 and 1.05 birds 10 ha^{-1} , respectively) and Ground Parrot (0.61 and 0.80 10 ha^{-1} , respectively). The Southern Emu-wren and Striated Fieldwren were detected across the full range of fire ages at both Lake Pedder and Lake St Clair (3-54 and 1-32 years post-fire, respectively), although were absent from some replicate sites within a few of the fire age classes. The only exception is that neither was visually detected at the one 12 year post-fire site (MCR) at Lake Pedder. However, a pair of Southern Emu-wrens were detected by call during one survey, indicating occasional use but not occupancy. The Ground Parrot appeared to have a more restricted distribution in relation to site fire ages at Lake Pedder and Lake St Clair (22-54 and 3-16 years post-fire, respectively), and as with the other residents was not detected at all replicate sites within these fire age classes. Overall, densities of the non-resident species were higher at Lake St Clair compared to Lake Pedder, particularly in the younger age classes (i.e. 1-3 years post-fire). Of these, the Tasmanian Thornbill, Crescent Honeyeater, and Superb Fairy-wren occurred at the most sites and had the highest overall densities at both locations (range 0.07-0.26 10 ha^{-1}). The Black Currawong, Dusky Robin, and Flame Robin were the only non-resident species included in the analyses that were only recorded at Lake St Clair; however, off-survey observations indicated that they were present within the Lake Pedder region. A number of the non-residents occurred across a broad range of fire ages at Lake Pedder and/or Lake St Clair, albeit inconsistently and generally at low densities, including the Tasmanian Thornbill, Superb Fairy-wren, Richard's Pipit, Black Currawong, Flame Robin, and Dusky Robin. However, the three Honeyeater species included in the analyses appeared to primarily occur in mature to old-growth moorlands and associated edge habitats at Lake Pedder and Lake St Clair (22-54 and 5-31 years post-fire, respectively; see Chapters 5-6).

The survey methods that were employed for this study were primarily intended to provide density estimates with low bias (Buckland *et al.* 2001); however, the moderate to large coefficients of variation (range CV = 28-128%; Table 4) and overlapping confidence

intervals for many of the species indicated the estimates only had moderate to low precision at the location scale. In the case of the distance density estimates, the variability in encounter rates and in the estimation of detection probability and cluster size all contributed to the overall variability. For the encounter rates, the small sample sizes and limited number of points/lines that could be established within many of the sites contributed to the high variability (i.e. on average 82% of total variance) (Fewster and Buckland 2004). Furthermore, no visual observations were recorded during any of the three seasons from 35% of the points that were surveyed ($n = 289$), indicating that the populations were patchily distributed both within and among sites. This was also reflected in the encounter rate variance as well as the estimated overdispersion parameters (range $b = 1-2.19$), which were nevertheless within the range identified as normal for patchily distributed biological populations (i.e. $1 \leq b \leq 4$, Burnham and Anderson 2002). Differences in detection probability were also a notable source of variance (i.e. on average 17% of total variance), while those in cluster sizes were only minor (i.e. on average 1% of total variance).

While the overall precision of the density estimates was lower than desirable, particularly at the site scale (not reported), as is often the case for such studies, these variance estimates provided a more honest and realistic assessment of uncertainty when compared to those calculated by simple indices of abundance that do not take such sources of variance into account (Casagrande and Beissinger 1997; Buckland *et al.* 2001; Burnham and Anderson 2002; Ellingson and Lukacs 2003; Diefenbach *et al.* 2007). These estimates of precision indicated that the buttongrass moorland avifauna exhibited a high degree of variability both within and among species, sites, and seasons; accordingly, these density estimates and subsequent analyses should be interpreted with a degree of caution.

Table 4. Summary of species occurrence and mean density estimates (\hat{D} , birds 10 ha⁻¹) from distance and VCP surveys conducted in buttongrass moorland sites at Lake Pedder ($n = 12$) and Lake St Clair ($n = 13$), Tasmania. Species included were present in $\geq 5\%$ of point/line transect surveys across the year ($n = 78$) (excluding aerial foragers/hunters). Densities for each location were calculated as the mean of site densities weighted by site area, including sites with zero density (Buckland *et al.* 2001). Site densities were derived from a mean of exact point and line transect survey estimates for all species except the Ground Parrot which was only based on the VCP surveys. Variance estimates were based on a Poisson distribution with species- and group-specific overdispersion factors (range $b = 1 - 2.19$) and include the percent coefficient of variance (CV) and lognormal 95% confidence intervals (CI). Species are listed in decreasing order of occurrence across locations. * Denotes moorland residents.

Species	Lake Pedder					Lake St Clair				
	Occurrence no. of 12 sites	3 - 54 yrs. post-fire	Mean \hat{D} (birds 10 ha ⁻¹)	CV %	Lognormal 95% CI	Occurrence no. of 13 sites	1 - 31 yrs. post-fire	Mean \hat{D} (birds 10 ha ⁻¹)	CV %	Lognormal 95% CI
Striated Fieldwren* <i>Calamanthus fuliginosus</i>	11	3 - 54	1.08	28.26	0.63 - 1.86	13	1 - 31	1.05	28.76	0.61 - 1.83
Southern Emu-wren* <i>Stipiturus malachurus</i>	12	3 - 54	1.83	49.10	0.74 - 4.55	10	1 - 31	1.64	50.32	0.65 - 4.15
Tasmanian Thornbill <i>Acanthiza ewingii</i>	6	3 - 31	0.10	44.64	0.05 - 0.24	7	2 - 31	0.16	30.88	0.09 - 0.29
Crescent Honeyeater <i>Phylidonyris pyrrhoptera</i>	7	22 - 54	0.18	35.70	0.09 - 0.35	5	8 - 31	0.07	45.49	0.03 - 0.17
Ground Parrot* <i>Pezoporus wallicus</i>	7	22 - 54	0.61	128.00	0.09 - 4.21	5	3 - 16	0.80	111.72	0.14 - 4.68
Superb Fairy-wren <i>Malurus cyaneus</i>	3	3 - 33	0.10	44.08	0.04 - 0.23	7	1 - 31	0.26	25.71	0.16 - 0.42
New Holland Honeyeater <i>Phylidonyris novaehollandiae</i>	5	22 - 54	0.10	44.64	0.05 - 0.24	2	14 - 31	0.02	82.46	0.00 - 0.07
Richard's Pipit <i>Anthus novaeseelandiae</i>	2	3 - 31	0.06	57.01	0.02 - 0.17	5	1 - 3	0.06	43.91	0.03 - 0.14
Yellow-throated Honeyeater <i>Lichenostomus flavicollis</i>	2	22 - 54	0.03	79.06	0.01 - 0.11	4	5 - 30	0.06	47.55	0.02 - 0.14
Green Rosella <i>Playcerus caledonicus</i>	4	25 - 27	0.03	79.06	0.01 - 0.12	1	5	0.01	108.03	0.00 - 0.06
Black Currawong <i>Strepera fuliginosa</i>	0	0	0	0	0	4	1 - 30	0.08	41.89	0.04 - 0.17
Flame Robin <i>Petroica phoenicea</i>	0	0	0	0	0	4	3 - 31	0.06	48.26	0.02 - 0.15
Dusky Robin <i>Melanodryas vittata</i>	0	0	0	0	0	3	1 - 30	0.07	44.16	0.03 - 0.15

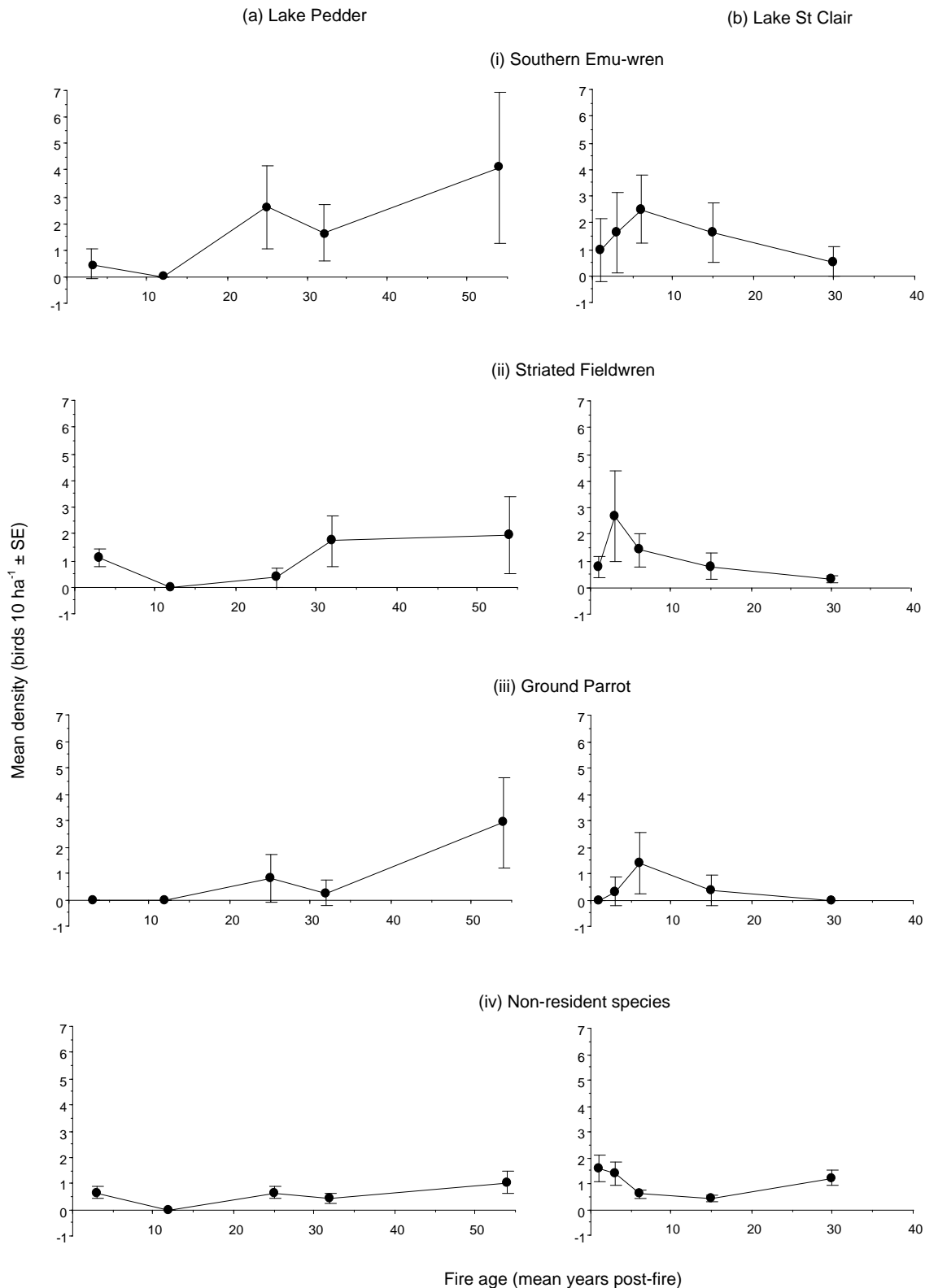


Fig. 4. Estimated mean densities (birds 10 ha⁻¹ ± SE) of the (i) Southern Emu-wren, (ii) Striated Fieldwren, (iii) Ground Parrot, and (iv) non-resident species in relation to mean years post-fire of sites within each fire age class in buttongrass moorlands at (a) Lake Pedder ($n = 12$; fire age classes = 3, 12, 22-27, 31-33, and 54 years post-fire) and (b) Lake St Clair ($n = 13$; fire age classes = 1, 2-3, 5-8, 14-16, and 30-31 years post-fire), Tasmania.

Species-habitat associations

Results from the univariate ordinary regression analyses of transformed mean avian densities in relation to six environmental variables are presented in Table 5 and trends in mean densities in relation to fire age classes are depicted in Figure 4. For simplicity, when densities are subsequently referred to in relation to the univariate and multivariate analyses, it is implied that they are transformed densities as specified in Table 5. Results from the hierarchical partitioning analyses depicting the percentage of explained variance for both independent and joint contributions from the full sets of environmental variables are presented in Figures 5 and 6.

Southern Emu-wren

There was a general trend in Southern Emu-wren densities to increase with time since fire at Lake Pedder (max. site 8.30 birds 10 ha⁻¹), while at Lake St Clair they initially increased up to approximately 5-8 years post-fire (max. site 4.69 birds 10 ha⁻¹) and then gradually decreased across the remainder of the chronosequence (Figure 4). Although fire age was not a significant factor for the Southern Emu-wren when all of the sites were included in the regression for Lake Pedder, an examination of residuals indicated that the site AIE had both high influence and leverage. This is largely due to the fact that AIE was included in the oldest fire age class at Lake Pedder and that the Southern Emu-wren was not identified by sight during any of the surveys and hence was recorded as a density of zero based on the exact distance analyses. However, results from the aural binomial surveys indicated that they did occur at the site throughout the study period, but for reasons explained earlier those data could not be incorporated into subsequent analyses. Accordingly, the regression was conducted with the 'false' absence at AIE omitted, which not only improved the fit of the correlation with fire age but also indicated that Southern Emu-wren densities significantly increased with time since fire at Lake Pedder, explaining 66.3% of variance compared to only 24.4% when the outlier AIE was included in the regression.

In contrast, there was no significant correlation between Southern Emu-wren densities and fire age at Lake St Clair. As shown in Chapter 3, there was a positive although nonlinear relationship between mean cover index and fire age at both Lake Pedder ($F = 66.063$; $df = 1, 9$; $P < 0.001$, $R^2 = 0.880$) and Lake St Clair ($F = 67.672$; $df = 1, 11$; $P < 0.001$, $R^2 = 0.860$). Consequently, there was also a significant positive correlation between Southern Emu-wren densities and cover at Lake Pedder (omitting AIE), explaining 39.6% of variance compared to 28.3% with AIE included, while there was no apparent relationship at Lake St Clair. There was also a significant negative correlation between densities and site area at Lake Pedder (omitting AIE), explaining 40.7% of variance compared to 18.9% with AIE included.

Finally, site riparian area was significantly and positively correlated with densities of the Southern Emu-wren at Lake St Clair, explaining 57.9% of variance.

In general, the results from the hierarchical partitioning for the Southern Emu-wren (Figure 5) were consistent with those from the univariate regressions. At Lake Pedder, the results including the outlier site AIE indicated that cover made the greatest independent contribution (29.1% explained variance), closely followed by fire age (26.2%), riparian area (21.1%), and site area (19.3%). However, when AIE was excluded from the analysis, it was apparent that the independent contribution from fire age amounted to more than twice that of the other variables (49.5%) and also made the highest joint contribution (37.9%). Site area (23.3%) and cover (20.2%) also made notable independent contributions, while that from riparian area was substantially reduced (3.6%). In contrast, at Lake St Clair site riparian area was clearly the most important variable (79.4%), while the other variables only had minor independent influence ($< 8\%$). Riparian area also had a much higher ratio of independent to joint effects than any of the other variables, suggesting that interactions with the other variables were relatively minor (Mac Nally 2000).

Striated Fieldwren

Overall, there was a moderate although inconsistent increase in Striated Fieldwren densities across the chronosequence at Lake Pedder (max. site $3.95 \text{ birds } 10 \text{ ha}^{-1}$). At Lake St Clair there was a marked increase in densities up to approximately 3 year post-fire (max. site $5.20 \text{ birds } 10 \text{ ha}^{-1}$) and then a gradual decrease across the chronosequence (Figure 4). Overall, this resulted in a significant negative correlation with fire age at Lake St Clair, explaining 41.8% of variance. Similar to the trend in relation to fire age, Striated Fieldwren densities at Lake St Clair were negatively correlated with cover; however, the relationship was relatively weak and non-significant, explaining only 25.3% of variance. Correlations between Striated Fieldwren densities and the remainder of the site variables were all non-significant, each explaining $< 12\%$ of variance.

In general, the results from the hierarchical partitioning for the Striated Fieldwren (Fig. 5) were consistent with those from the univariate regressions. Although there were no significant results in relation to the variables at Lake Pedder, the hierarchical partitioning indicated that fire age made substantially greater independent (44.5%) contributions than the other variables, followed by site area (24.0%), while the other variables independently each only explained $< 18\%$ of the total variance. The high joint contributions from both fire age (44.0%) and site area (61.5%) suggested complex interrelationships with each other and the other variables. At Lake St Clair, fire age provided the greatest independent (54.9%) contribution, followed by cover (25.2%), while the other variables independently each only

explained < 11% of the total variance. The high joint contributions from both fire age (42.8%) and cover (36.2%) suggested complex interrelationships with each other and the other variables. Although the trends in densities as indicated by the univariate regressions are opposite at Lake Pedder (positive) and Lake St Clair (negative), these results suggested that fire age was the most important explanatory variable in relation to Striated Fieldwren densities in buttongrass moorlands. Due to the lack of a strong correlation between densities and fire age at Lake Pedder ($R^2 = 0.100$), it was difficult to determine the actual direction of the trend, if any exists.

Ground Parrot

Results from the univariate logistic regressions in relation to Ground Parrot occurrence are presented in Table 6. At Lake Pedder, the probability of occurrence of Ground Parrots was significantly and positively correlated with both fire age and cover, explaining 60.5% and 79.2% of variance, respectively. The results showed that Ground Parrots were more likely to occur at older sites with higher cover values at Lake Pedder. In contrast, at Lake St Clair there was a relatively weak and non-significant negative correlation with fire age, explaining only 27.8% of variance, and no apparent association with cover. The Ground Parrot was the only species for which edge contrast index was a significant factor. Similar to the analyses for the Southern Emu-wren, one site (MCR) was identified as an outlier. This was largely due to the fact that it was established as a relatively small research burn plot (21.22 ha) (Marsden-Smedley 1993) within a much larger patch of moorland and thus had a very low edge contrast index. Ground Parrots were not recorded there during any of the surveys, although one off-survey observation indicated that some individuals may periodically use the site. Accordingly, the regression was conducted with the outlier MCR omitted, which not only improved the fit of the relationship but also indicated that the probability of Ground Parrot occurrence at Lake Pedder significantly decreased at sites with high contrast edges (e.g. a high proportion of forest edges), explaining 51.4% of variance compared to only 11.7% when the outlier MCR was included in the regression. These results indicated that despite its low edge contrast index, MCR apparently did not provide good habitat relative to portions of the surrounding moorlands due to its younger age and lower cover. At Lake St Clair there was also a significant negative correlation between Ground Parrot occurrence and edge contrast index, explaining 64.2% of variance. At Lake St Clair there was a significant positive correlation between the probability of occurrence of Ground Parrots and the percentage of suitable habitat found within a 400 m radius of the survey point at each site (~ 50 ha), explaining 57.1% of variance. At Lake Pedder there was also a positive but non-significant correlation ($P = 0.053$), explaining 36.0% of variance; however, at either location Ground Parrots did not occur at sites with < 45% of suitable habitat within the 50 ha circles.

Results from the ordinary regressions in relation to Ground Parrot density were generally consistent with those obtained above in relation to occurrence, largely due to the necessity to include sites with no detections in the density dataset (see Methods). The Ground Parrot was absent from sites < 13 years post-fire at Lake Pedder and then showed a general increase in densities across the chronosequence (max. site 4.19 birds 10 ha⁻¹) (Figure 4). At Lake St Clair, it was absent from the one recently burnt site, gradually increased in density up to approximately 5-8 years post-fire (max. site 3.19 birds 10 ha⁻¹), and then gradually decreased in density and was absent from the oldest sites. Similar to the above, there were significant positive correlations between Ground Parrot densities and both fire age and cover at Lake Pedder, explaining 56.9% and 48.9% of variance, respectively, and no significant relationships with fire age and cover at Lake St Clair. Although Ground Parrot occurrence did not appear to be influenced by site area, there was a significant negative correlation with density and site area at Lake Pedder, explaining 37.4% of variance. The trend at Lake St Clair was opposite, with densities increasing with site area, explaining 35.9% of variance. Similar to the analyses above, there were significant negative correlations between Ground Parrot densities and edge contrast index at Lake Pedder (excluding the high influence and leverage MCR) and Lake St Clair, explaining 58.1% and 36.2% of variance, respectively. Finally, the percentage of suitable habitat was also a significant factor, with densities being higher at sites surrounded by a greater proportion of suitable habitat at Lake St Clair, explaining 45.9% of variance, while at Lake Pedder the sites were typically found within large, contiguous patches of moorland and hence there was a weak, non-significant correlation, explaining only 20.5% of variance. These percentages of explained variance in relation to density were likely to be lower than those in relation to occurrence since one would expect that the proportions of suitable habitat beyond the sites and/or home ranges of the more mobile Ground Parrots (i.e. at the landscape scale) would have a greater influence on dispersal and hence occupancy than densities *per se* (i.e. at the site scale).

In general, the results from the hierarchical partitioning for the Ground Parrot (Fig. 6) were consistent with those from the univariate regressions. Since the results between the occurrence and density analyses were comparable, only the hierarchical partitioning graphs in relation to Ground Parrot densities are presented. At Lake Pedder, the results including the outlier site MCR indicated that fire age had the greatest independent contribution (37.1%), followed by cover (22.7%), while those from the remainder of the variables were all < 18%. However, when MCR was excluded from the analysis it was apparent that the independent contribution from fire age was reduced (25.7%) while that from edge contrast index was increased (28.0%), but those from the remainder were again < 18% and had similar joint contributions (11-24%). At Lake St Clair, the percentage of suitable habitat had the greatest independent (27.1%) and joint (41.8%) contribution, closely followed by the

independent contributions from edge contrast index (24.8%), area (21.8%), and fire age (18.8%). The independent contribution from cover was only minor (7.6%) and offset by negative joint effects (-7.5%), suggesting that the other variables may have been suppressing the independent influence of cover (Chevan and Sutherland 1991; Mac Nally 1996b).

Non-resident species

There were no significant relationships between non-resident densities and fire age at either Lake Pedder or Lake St Clair. The only notable pattern was a peak in non-resident mean densities at the youngest site at Lake St Clair (i.e. 1 year post-fire), as well as a secondary peak at the oldest sites (i.e. 30-31 years post-fire), which were the only two age classes at either location in which mean densities exceeded those of all three resident species (Figure 4). The sole variable significantly correlated with non-resident densities was site area at Lake St Clair, indicating decreasing densities with increasing site area and explaining 65.1% of variance. Correlations between non-resident densities and the remainder of the site variables were all non-significant, each explaining < 13% of variance.

In general, the results from the hierarchical partitioning for the non-resident species (Figure 6) were consistent with those from the univariate regressions. Although there were no significant results in relation to the variables at Lake Pedder, the hierarchical partitioning indicated that riparian area was the strongest independent correlate (36.4%), consistent with their strong selection for riparian habitats compared to the matrix, as described in Chapter 5. However, this was largely offset by joint negative interactions with the other variables (-25.3%), suggesting that they may have been suppressing the independent influence of riparian area. A similar pattern in independent contribution being offset by negative joint interactions was also apparent for cover. Large positive joint contributions for fire age (52.4%) and area (79.0%) suggested they had some influence on densities; however, overall the results suggested that none of the variables had a strong independent influence on non-resident densities, while the other variables had little independent and joint effects. At Lake Pedder, site area clearly had the strongest independent (69.8%) and joint (151.3%) contribution towards the overall variance, while those from the other variables were relatively negligible. This strong contribution may have reflected the fact that they could be more readily detected at smaller sites since a greater proportion of their preferred edge habitats (see Chapter 5) were closer to the transects when compared to larger sites where there was a negative bias towards detecting non-residents from the interior of the sites, as well as a preference for such sites with a higher edge to area ratio. Furthermore, all sites at Lake St Clair had higher edge contrast indices than any at Lake Pedder, which may have accounted for the greater contribution of area towards variance at Lake St Clair versus Lake Pedder. However, as stated above and indicated by the high joint contribution, area may be

confounded with the influence of the other variables as well as its influence on the calculation of bird densities.

Table 5. Ordinary linear regression results from univariate tests of (transformed) mean densities (birds ha⁻¹) of the (√) Southern Emu-wren, (√) Striated Fieldwren, (log₁₀(y + 0.001)) Ground Parrot, and (√) non-resident species across three seasons in relation to site fire age (years post-fire), mean cover index, area (ha), edge contrast index (ECI, %), riparian area (%), and suitable Ground Parrot habitat within a 400 m radius of the survey point (%), in buttongrass moorlands at Lake Pedder (*n* = 12) and Lake St Clair (*n* = 13), Tasmania. * Denotes significance at the 0.05 level. ^ Analyses excluding outlier sites (AIE for Southern Emu-wren and MCR for Ground Parrot).

Species	Variable	Lake Pedder						Lake St Clair					
		Estimate	SE	df	<i>F</i>	<i>P</i>	<i>R</i> ²	Estimate	SE	df	<i>F</i>	<i>P</i>	<i>R</i> ²
Southern Emu-wren	Fire age	0.009	0.005	1,10	3.220	0.103	0.244	-0.005	0.007	1,11	0.617	0.449	0.053
	Fire age ^	0.014	0.003	1,9	17.675	0.002*	0.663						
	Cover	0.005	0.003	1,10	3.947	0.075	0.283	-0.001	0.003	1,11	0.090	0.770	0.008
	Cover ^	0.006	0.002	1,9	5.898	0.038*	0.396						
	Area	-0.010	0.006	1,10	2.330	0.158	0.189	0.003	0.003	1,11	0.780	0.396	0.066
	Area ^	-0.014	0.005	1,9	6.190	0.035*	0.407						
	ECI	0.000	0.008	1,10	0.004	0.951	0.000	-0.008	0.009	1,11	0.875	0.370	0.074
Striated Fieldwren	Riparian	0.035	0.031	1,10	1.339	0.274	0.118	0.089	0.023	1,11	15.099	0.003*	0.579
	Fire age	0.004	0.003	1,10	1.110	0.317	0.100	-0.011	0.004	1,11	7.909	0.017*	0.418
	Cover	0.001	0.002	1,10	0.428	0.528	0.041	-0.003	0.002	1,11	3.731	0.080	0.253
	Area	0.000	0.005	1,10	0.000	0.993	0.000	0.000	0.003	1,11	0.003	0.954	0.000
	ECI	0.004	0.005	1,10	0.690	0.426	0.065	-0.008	0.006	1,11	1.500	0.246	0.120
Ground Parrot	Riparian	0.009	0.021	1,10	0.175	0.685	0.017	0.019	0.025	1,11	0.594	0.457	0.051
	Fire age	0.046	0.013	1,10	13.219	0.005*	0.569	-0.038	0.025	1,11	2.315	0.156	0.174
	Cover	0.024	0.008	1,10	9.555	0.011*	0.489	-0.004	0.011	1,11	0.147	0.708	0.013
	Area	-0.048	0.020	1,10	5.976	0.035*	0.374	0.028	0.011	1,11	6.160	0.030*	0.359
	ECI	-0.036	0.025	1,10	1.996	0.188	0.166	-0.073	0.029	1,11	6.246	0.030*	0.362
	ECI ^	-0.076	0.022	1,9	12.454	0.006*	0.581						
Non-resident species	Suitable habitat	3.830	2.384	1,10	2.581	0.139	0.205	3.818	1.250	1,11	9.331	0.011*	0.459
	Fire age	0.001	0.002	1,10	0.086	0.776	0.009	-0.001	0.003	1,11	0.024	0.879	0.002
	Cover	0.001	0.001	1,10	0.492	0.499	0.047	-0.002	0.001	1,11	1.703	0.219	0.134
	Area	0.000	0.003	1,10	0.012	0.915	0.001	-0.005	0.001	1,11	20.556	0.001*	0.651
	ECI	0.001	0.003	1,10	0.066	0.802	0.007	0.003	0.004	1,11	0.331	0.576	0.029
	Riparian	0.012	0.013	1,10	0.861	0.375	0.079	-0.004	0.017	1,11	0.045	0.837	0.004

Table 6. Logistic regression results from univariate tests of Ground Parrot occurrence across three seasons in relation to environmental variables (Table 5) in buttongrass moorlands at Lake Pedder ($n = 12$) and Lake St Clair ($n = 13$), Tasmania. * Denotes significance at the 0.05 level. ^ Analyses excluding outlier site (MCR).

Variable by location	Estimate	SE	df	Log-likelihood χ^2	P	Nagelkerke pseudo- R^2	CI (95%) for odds ratio
Lake Pedder							
Fire age	0.154	0.098	1	7.165	0.007*	0.605	0.964 - 1.412
Cover	0.223	0.197	1	10.642	0.001*	0.792	0.848 - 1.840
Area	-0.092	0.058	1	3.452	0.063	0.336	0.814 - 1.021
ECI	-0.053	0.053	1	1.089	0.297	0.117	0.854 - 1.053
ECI ^	-0.178	0.103	1	5.183	0.023*	0.514	0.684 - 1.025
Suitable habitat	11.230	7.234	1	3.736	0.053	0.360	0.052 - 1 E+11
Lake St Clair							
Fire age	-0.115	0.083	1	2.978	0.084	0.278	0.758 - 1.048
Cover	-0.016	0.022	1	0.546	0.460	0.056	0.943 - 1.027
Area	0.055	0.036	1	3.255	0.071	0.301	0.984 - 1.134
ECI	-0.379	0.238	1	8.326	0.004*	0.642	0.429 - 1.091
Suitable habitat	22.669	20.817	1	7.096	0.008*	0.571	0.000 - 4 E+27

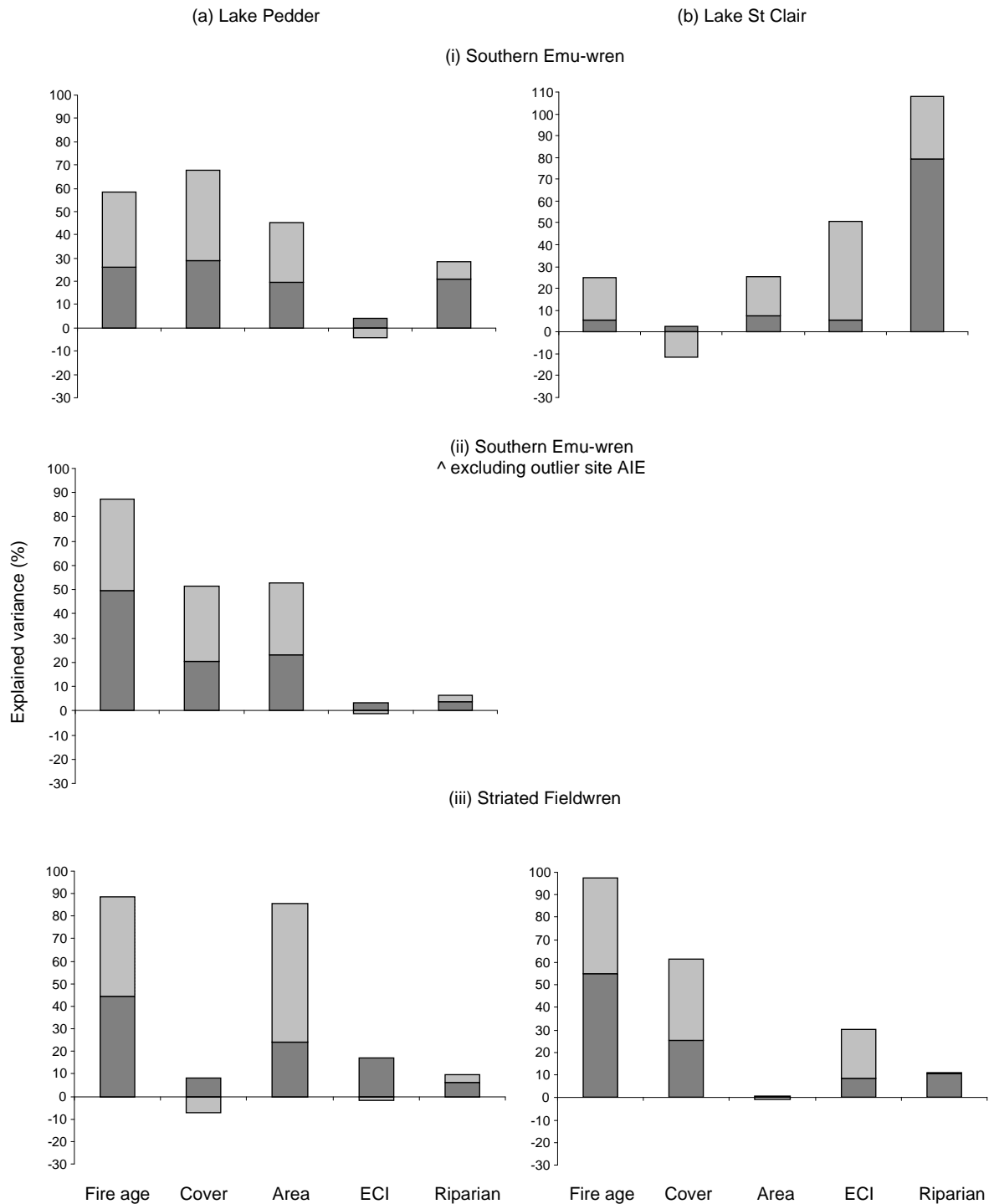


Fig. 5. Percentages of explained independent (■) and joint (□) variance for five explanatory environmental variables (Table 5) related to (transformed) mean densities (birds ha⁻¹) of the (i) (✓) Southern Emu-wren, (ii) Southern Emu-wren ^ excluding outlier site AIE, and (iii) (✓) Striated Fieldwren, across three seasons in buttongrass moorlands at (a) Lake Pedder ($n = 12$) and (b) Lake St Clair ($n = 13$), Tasmania as determined from hierarchical partitioning. The total explained variance for each variable is equivalent to the percentage of univariate R^2 and indicates the relative influence of each variable on transformed densities. Note that y-axis values differ for (i)(b).

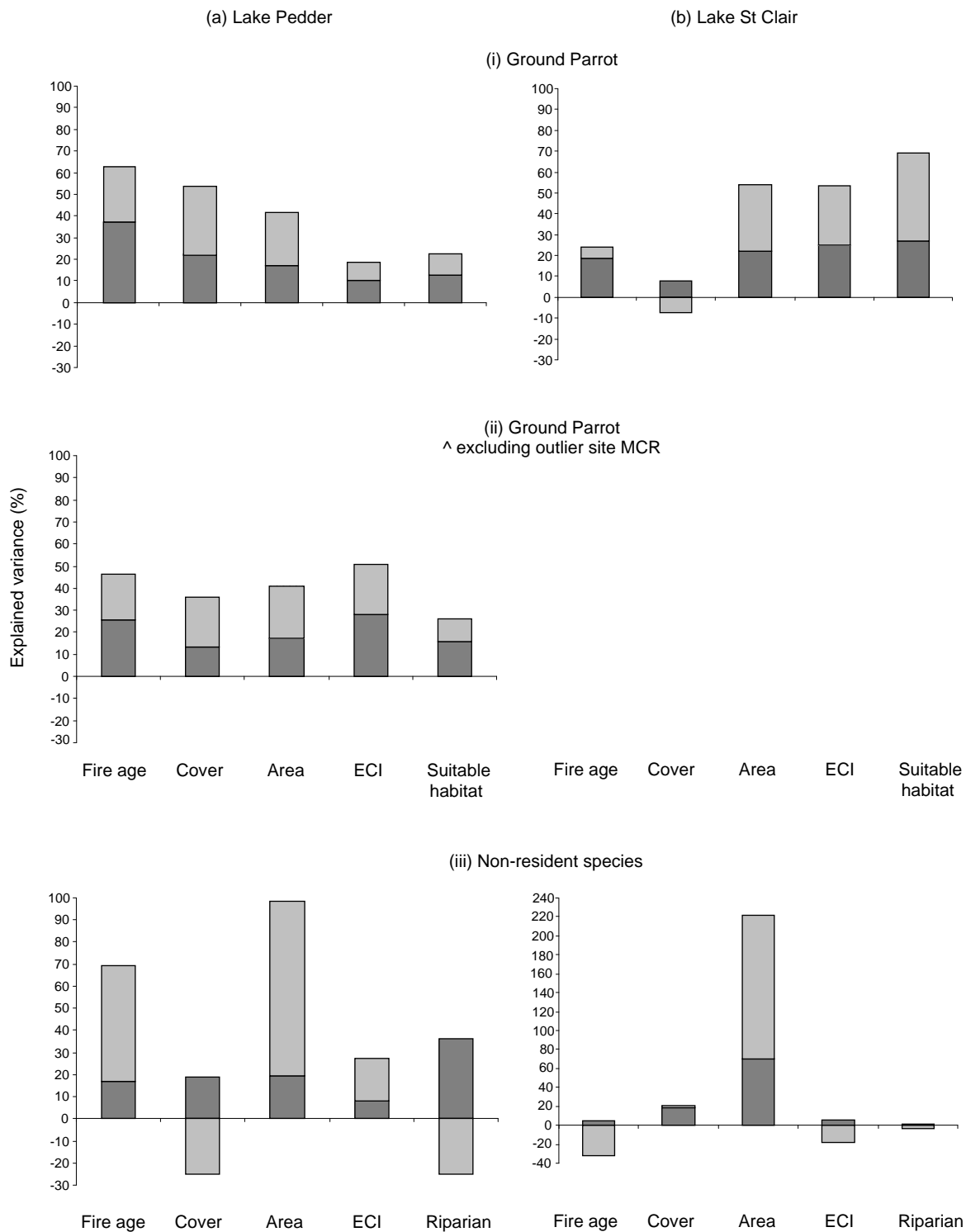


Fig. 6. Percentages of explained independent (■) and joint (□) variance for five explanatory environmental variables (Table 5) related to (transformed) mean densities (birds ha⁻¹) of the (i) ((log₁₀(y + 0.001)) Ground Parrot, (ii) Ground Parrot ^ excluding outlier site MCR, and (iii) (√) non-resident species, across three seasons in buttongrass moorlands at (a) Lake Pedder (*n* = 12) and (b) Lake St Clair (*n* = 13), Tasmania as determined from hierarchical partitioning. The total explained variance for each variable is equivalent to the percentage of univariate *R*² and indicates the relative influence of each variable on transformed densities. Note that y-axis values differ for (iii)(b).

Avifaunal habitat associations

The RELATE analyses presented in Table 7 yielded significant correlations between the similarity matrix of square-root transformed avifaunal mean densities and composition and that of fire age at both Lake Pedder and Lake St Clair ($r_s = 0.318$ and 0.238 ; $P = 0.023$ and 0.037 , respectively), while area was also a significant factor at Lake Pedder ($r_s = 0.420$; $P = 0.008$). Although these results are significant, the magnitude of the correlation coefficients indicated only moderate correspondence between the avifaunal and specified environmental matrices taken singly. The BIO-ENV analyses (Table 7) indicated that the rank correlation was maximised with a combination of the fire age, cover, area, and edge contrast index variables at Lake Pedder ($r_s = 0.456$), while at Lake St Clair only the fire age and riparian area variables were selected ($r_s = 0.408$). These moderate correlations indicated that the subset of selected variables in combination still only explained less than half of the variance in patterns of avifaunal densities and composition within each location. Overall, fire age was inferred to be the most influential variable since it was significantly correlated with avifaunal patterns at both locations as indicated by the RELATE procedure, and was selected in the majority of the best matches as indicated by the BIO-ENV procedure. These results were in general agreement with those from the univariate regressions and multivariate hierarchical partitioning and provided additional support that, of the variables included in the analyses, fire age appeared to have the strongest association with avifaunal composition and densities.

Results from the SIMPER analysis presented in Table 8 showed mean one-way similarities of avifaunal patterns in square-root transformed densities and composition of 46.90 at Lake Pedder and 38.15 at Lake St Clair. At Lake Pedder, the resident Southern Emu-wren, Striated Fieldwren, and Ground Parrot dominated the community structure, together contributing 84% of similarity between sites. Contributions from the remaining species were relatively low and except for the Crescent Honeyeater were $< 5\%$. At Lake St Clair, the residents were somewhat less dominant, together contributing 65% of similarity between sites, while the Tasmanian Thornbill and Superb Fairy-wren were also influential, together contributing 23% towards inter-site similarities. Contributions from the remaining species were all $< 5\%$. Despite the use of square-root transformed densities to reduce the influence of the three resident species in the RELATE and BIO-ENV analyses, it was still apparent that the resident species together with only three non-resident species had the greatest influence on avian relationships at the community level. Furthermore, high ratios of the mean contribution of similarity of each species to their standard deviation indicate which species typify sites (Clarke and Warwick 1994). Although the ratios for the Striated Fieldwren were just below the typical threshold of 1.4 for similarities within locations, they indicated the relative importance it played in typifying the moorland avifauna as compared to

the other resident and non-resident species which were less consistently abundant within each location.

Table 7. Relationships between similarity matrix of avifaunal composition and mean densities ($\sqrt{\text{birds ha}^{-1}}$) and those for each environmental variable, including site fire age (years post-fire), mean cover index, area (ha), edge contrast index (ECI, %), and riparian area (%), using the Spearman rank correlation coefficient (r_s) from the RELATE procedure and the subset of variables that maximised the rank correlation value from the BIO-ENV procedure for buttongrass moorland sites at Lake Pedder ($n = 11$, excluding MCR) and (b) Lake St Clair ($n = 13$), Tasmania. * Denotes significance at the 0.05 level.

Variable by location	RELATE			BIO-ENV
	r_s	Permutations (of 999) with scores $\geq r_s$	P	Maximum r_s and selected variables (+)
Lake Pedder				0.456
Fire age	0.318	22	0.023*	+
Cover	0.347	62	0.063	+
Area	0.420	7	0.008*	+
ECI	0.168	94	0.095	+
Riparian	-0.004	464	0.465	
Lake St Clair				0.408
Fire age	0.238	36	0.037*	+
Cover	-0.007	442	0.443	
Area	-0.028	507	0.508	
ECI	0.034	352	0.353	
Riparian	0.209	120	0.121	+

Table 8. Species contributions to mean one-way similarities of avifaunal patterns in composition and mean densities ($\sqrt{\text{birds ha}^{-1}}$) from the SIMPER procedure (cut-off = 90%) within buttongrass moorland sites at Lake Pedder and Lake St Clair, Tasmania.

Species by location	Mean similarity	Similarity/ SD	Contribution (%)	Cumulative contribution (%)
Lake Pedder				
Southern Emu-wren	19.14	1.24	40.80	40.80
Striated Fieldwren	14.15	1.32	30.17	70.98
Ground Parrot	6.11	0.70	13.02	84.00
Crescent Honeyeater	3.82	0.73	8.15	92.14
Lake St Clair				
Striated Fieldwren	14.13	1.32	37.05	37.05
Southern Emu-wren	8.81	0.68	23.10	60.14
Tasmanian Thornbill	4.69	0.56	12.30	72.44
Superb Fairy-wren	3.98	0.52	10.44	82.89
Ground Parrot	2.16	0.35	5.65	88.54
Crescent Honeyeater	1.49	0.37	3.90	92.44

Discussion

Overall patterns in avifaunal occurrence and densities

This is the first comprehensive and quantitative study of the avifauna of Tasmanian buttongrass moorlands. During the course of surveys, a total of only 25 species were recorded, in addition to unspecified raptor species, and amounted to 45% of the 55 species reported in the literature as having been observed in buttongrass moorlands and associated habitats in Tasmania (see Table 1, Chapter 1). Based on occurrence and relative density data, this study provides the first quantitative evidence that the core avian community of buttongrass moorlands is composed of only three year-round resident species within the study area that depend exclusively on moorlands for breeding, feeding, and other resource needs, namely the Southern Emu-wren, Striated Fieldwren, and Ground Parrot. Only 12 other species, in addition to unspecified raptors, occurred in more than 5% of all point/line transect surveys. Since none of these species were continuously present and many of them occurred in relatively low densities and were primarily associated with adjacent ecotonal, scrub, woodland, and forest habitats (Watts 2002; see Chapter 5), they should be regarded as marginal or periodic users of moorlands within the study area. The remaining 10 species were rarely observed and are not species typically expected in moorlands. Thus, the results from this study generally concur with previous qualitative accounts (e.g. Ridpath and Moreau 1966; Brown *et al.* 1993; Driessen 2006; see Chapter 1) and demonstrate that the avifauna of Tasmanian buttongrass moorlands has a relatively low species richness characterised by only three year-round resident specialist species and approximately 12 non-resident species in the study area, including those listed in Table 4 and the Welcome Swallow and Tree Martin, both common summer migrants (pers. obs.; see Chapter 1). However, the actual number of species and the community composition likely varies with geography, site attributes, season, and the precise definitions of ‘moorland’ habitat and ‘characteristic’ species that have been used by various authors (P. Brown pers. comm. 2005; see Chapter 1). For the purposes of this discussion, inferences from the distance sampling surveys are limited to the study area, including sites at Lake Pedder and Lake St Clair, and should not be extrapolated to the landscape as a whole (Buckland *et al.* 2008).

Overall, the observed patterns of avifaunal occurrence and density across the space-for-time fire chronosequences at Lake Pedder and Lake St Clair were complex and revealed high levels of variability within and among species, sites, and locations. Dissimilarities in composition and densities were partially explained by differences in location (i.e. moorland type, soil productivity, and altitude), fire age, cover, and habitat composition and configuration. Measures of avian richness and density in particular are often implicitly assumed to be positively correlated with habitat quality *per se*. Habitat quality is defined as the relative suitability of an area to support a reproducing population of a given resident

species (i.e. carrying capacity) (Maurer 1986; Hobbs and Hanley 1990) and quantified as the product of density, survival, and reproductive success (Van Horne 1983). However, this assumption has often not been fully validated since it is necessary to obtain detailed demographic data (e.g. survival, reproduction, dispersal) from the population(s) of interest to make reliable inferences for management (Van Horne 1983; Hobbs and Hanley 1990; Wheatley *et al.* 2002). In particular, species-specific factors such as territoriality, fecundity, dispersal, and niche breadth, as well as environmental factors such as temporal (e.g. disturbance related time-lags) and spatial (e.g. patchiness) heterogeneity of resources may result in what are deemed low- or moderate-quality habitats supporting higher densities than those considered high-quality habitats (Fretwell and Lucas 1969; Van Horne 1983; Vickery *et al.* 1992; Hobbs and Hanley 1990; Vierling 1999; Wheatley *et al.* 2002). Furthermore, in dynamic landscapes such as within the study area, the complex interrelationships of source and sink habitats in both space and time make it even more difficult to solely explain the observed patterns in densities with site-level attributes (Dunning *et al.* 1992; Brawn and Robinson 1996). As was the case with this study, detailed demographic data are rarely available due to the resource intensive nature of such studies, particularly for cryptic species such as the moorland residents. In the absence of such data and for the purpose of this discussion, it is necessary to assume that there is an as yet undetermined, although presumably positive, relationship between species densities and habitat suitability. A literature review by Bock and Jones (2004) indicated that this may be a reasonable assumption in most cases. If detailed demographic and multi-year survey data become available in the future, the results from this study should be reassessed in order to elucidate the nature of these relationships and their underlying processes, particularly in relation to the resident species. Given this limitation, it is currently difficult to make any definitive statements concerning the habitat relationships of the moorland avifauna. Nevertheless, some important patterns have emerged that provide a better understanding of this previously little studied avian community and represents an important first step for more informed management of buttongrass moorland ecosystems and future research on the avifauna.

The resident Southern Emu-wren, Striated Fieldwren, and Ground Parrot dominated the avifauna in terms of site occupancy, estimated densities, and inter-site similarities in community structure at Lake Pedder and Lake St Clair, consistent with expectations based on previous accounts of the moorland avifauna (see Table 1, Chapter 1). Both the Striated Fieldwren and Southern Emu-wren were nearly ubiquitous throughout the study area, while the Ground Parrot was patchily distributed and occurred at just under half of the study sites. The only non-resident species that notably contributed (i.e. > 5%) to avian relationships at the community level are relatively common and widespread species (Watts 2002; pers. obs.), including the Crescent Honeyeater at Lake Pedder and the Tasmanian Thornbill and Superb

Fairy-wren at Lake St Clair. The remaining non-resident species occurred inconsistently across the study area and were primarily associated with adjacent woodland and forest habitats (see Chapter 5).

As with the avifaunas of other pyrogenic vegetation communities in Australia and abroad (for reviews see Woinarski 1999a, 1999b; Saab and Powell 2005; and Chapter 7), the results from this study suggest that fire is one of the primary factors that influence both the composition and densities of bird species in Tasmanian buttongrass moorlands over the course of successional time scales (i.e. ~ 1 to > 50 years post-fire). In particular, fire age was identified as a significant independent factor in relation to the densities of the three resident species at Lake Pedder and/or Lake St Clair, and in maximising rank correlations of similarity matrices at the community level at both locations. In addition, there were notable joint interactions between fire age and the other explanatory variables discussed below, suggesting that some of the inter-site variability could be attributed to complex interactions between the habitat variables included in the analyses. These results, along with those from Chapters 5-6, are more thoroughly discussed in Chapter 7 in relation to similarly depauperate sedgeland and heathland communities within a national and global context, as well as models of post-fire faunal succession within a theoretical context.

Species occurrence in relation to post-fire succession

Overall, patterns in species occurrence in relation to time since last fire appeared to reflect the relative suitability of different seral stages of post-fire vegetation in relation to the specific niche breadths and habitat preferences of the resident and non-resident species in the study area (see Chapters 1 and 3). The Southern Emu-wren and Striated Fieldwren were detected across the full range of fire ages at both locations (3-54 at Lake Pedder, 1-32 years post-fire at Lake St Clair; except MCR for the former, see below). Although to date no formal studies have been conducted on either species in Tasmanian moorlands, the Lyell Highway Fire Management Plan and references therein (PWS 1996, citing Gellie 1980 and Brown *et al.* 1993) provided limited and principally qualitative information. In contrast to the results from this study, it suggested that the Southern Emu-wren does not inhabit sites < 9 years post-fire, while it reported that it was unknown whether it inhabited sites > 15 years post-fire. In the case of the Striated Fieldwren, it reported that it was known to inhabit sites 3-11 years post-fire, while it was unknown whether it inhabited younger or older sites. Based on the present study, it is apparent that both younger and older sites can provide suitable resources to enable use and/or occupancy by the Southern Emu-wren and Striated Fieldwren.

The Ground Parrot appeared to have a more restricted distribution in relation to site fire ages when compared to the other resident species (22-54 at Lake Pedder, 3-16 years post-fire at Lake St Clair). While the Ground Parrot was absent at sites < 22 years post-fire at Lake Pedder and < 3 and > 16 years post-fire at Lake St Clair, previous research on the Ground Parrot in Tasmanian moorlands by Bryant (1991, 1992, 1994) identified Ground Parrots across the full range of sites investigated in Tasmania (including blanket and eastern moorlands), from 1-90 years post-fire by flushing and from 2 months to > 30 years post-fire by call surveys. However, due to the fact that extensive surveys were conducted throughout the state for her study, it is not surprising that the results from the present geographically limited study indicated a more restricted occupancy of sites in relation to time since fire at both Lake Pedder and Lake St Clair. It should be noted that no species were visually detected at the one 12 year post-fire site (MCR) at Lake Pedder during the survey. This site was considered to be an outlier since it was a relatively small research burn plot adjacent to older moorlands with greater cover (see Chapter 2), and thus presumably provided a less suitable habitat for the cover-dependent resident species. Nevertheless, both on-survey auditory and off-survey visual detections of the Southern Emu-wren and Ground Parrot, respectively, indicated occasional use, but not occupancy, of this site. Due to the lack of availability and hence replication of sites between 3 and 22 years post-fire at Lake Pedder, it is difficult to make any definitive inferences regarding possible trends between these fire ages in blanket moorlands for any of the species. In particular, in the case of the Ground Parrot, the limited representation of younger sites coupled with the slower rate of post-fire vegetative recovery of these low productivity sites may have resulted in some bias towards older blanket moorlands relative to patterns of occupancy in a statewide context (Bryant 1991).

Among the non-resident species, the Tasmanian Thornbill, Superb Fairy-wren, Richard's Pipit, Black Currawong, Flame Robin, and Dusky Robin occurred across a broad range of fire ages at Lake Pedder and/or Lake St Clair, albeit inconsistently. However, the Crescent, New Holland, and Yellow-throated Honeyeater species included in the analyses appeared to primarily occur in mature to old-growth moorlands and associated edge habitats at Lake Pedder and Lake St Clair (i.e. 22-54 and 5-31 years post-fire, respectively; see Chapter 5). In contrast, results presented in Chapter 6 demonstrated that all three species utilised recently burnt moorlands and that short-term densities may increase under some conditions. It should be noted that the necessity of pooling the non-resident species into a single class may have masked more definitive species-specific patterns of occurrence in relation to fire age.

Species densities in relation to post-fire succession

All three resident species demonstrated qualitatively similar, although variable, patterns in post-fire changes in densities. The Southern Emu-wren had the highest estimated mean densities at both Lake Pedder and Lake St Clair (1.83 and 1.64 birds 10 ha⁻¹, respectively), followed by the Striated Fieldwren (1.08 and 1.05 birds 10 ha⁻¹, respectively), and Ground Parrot (0.61 and 0.80 10 ha⁻¹, respectively). Intra-specific mean densities and inter-specific ranks of densities across fire ages were remarkably similar between locations. These densities are within the range of those reported in previous studies conducted in Tasmania and/or on the mainland for the Southern Emu-wren, Striated Fieldwren, and Ground Parrot, despite considerable differences in survey methods, analyses, and environmental variables in comparison to this study (Table 9).

Table 9. Comparison of density ranges (birds 10 ha⁻¹) of resident species of Tasmanian buttongrass moorlands reported in previous studies conducted in Australia.

Species	Density range (birds 10 ha ⁻¹)	Habitat, State, Reference
Southern Emu-wren	0.4-45.0	Heathland, NSW, Jordan 1987a; dry and wet heathland, QLD, McFarland 1988b, 1994; sedgeland, shrubland, banksia-eucalypt forest, NSW, Wood 1995; low heathland, NSW, Gosper and Baker 1997; heathland, VIC, Maguire 2006b
Striated Fieldwren	0.6-9.0	Samphire shrubland, VIC, Watson 1955; low heathland, NSW, Gosper and Baker 1997; regrowth dry sclerophyll forest, TAS, Taylor <i>et al.</i> 1997
Ground Parrot	0.2-6.5	Sedgeland and heathland, VIC, Meredith and Isles 1980; heathland, NSW, Jordan 1987b; dry and wet heathland, QLD, McFarland 1988b, 1991c; heathland, WA, Burbidge 1989; sedgeland and graminoid heathland (i.e. buttongrass moorland), TAS, Bryant 1991

The lowest mean densities of the residents generally occurred within the youngest sites surveyed at Lake Pedder and Lake St Clair (i.e. 3 and 1 years post-fire, respectively). At Lake Pedder, the resident species generally increased in densities across the chronosequence, with the highest mean densities at the oldest sites surveyed (i.e. 54 years post-fire), which for the Striated Fieldwren were at least two times the densities at the youngest sites, and for the Southern Emu-wren up to over ten times. At Lake St Clair, the mean densities of the residents peaked between 2-8 years post-fire, after which they all gradually declined to their lowest densities at the oldest sites surveyed (i.e. 30-31 years post-fire). For the Ground Parrot, the peaks in densities ranged from at least three times those at the oldest sites, and for the Striated Fieldwren they were up to over ten times. Despite the apparent trends in mean densities in relation to fire age (Figure 4), they were only statistically significant within a linear model for the Southern Emu-wren and Ground Parrot at Lake Pedder (positive) and the Striated Fieldwren at Lake St Clair (negative). Species-specific comparisons between

this study and previous research provide a broader context in which to assess these results and are discussed below.

Although this is the first study to investigate the influence of fire on Southern Emu-wrens in Tasmanian moorlands, studies conducted on the mainland have shown some comparable post-fire trends in occupancy and densities (see Chapter 1). Studies to date likewise indicate that Southern Emu-wrens were either absent or present in substantially reduced numbers soon after disturbance by fire and did not start to increase until one to a few years post-fire (Jordan 1987a; McFarland 1988b, 1994; Reilly 1991a; Loyn 1997). The relatively fragmented nature of their habitats on the mainland may pose additional issues in relation to post-fire recolonisation due to the limited dispersal capabilities of Southern Emu-wrens. This is particularly the case for extensive wildfires, since the longest recorded movements to date are only 1.2 km for juveniles (Maguire and Mulder 2004) and approximately 2.5 km for adults (Pickett 2000). Although the trend of Southern Emu-wren densities in relation to fire age at Lake St Clair was not significant, data indicated initially low post-fire densities, followed by a rapid increase and peak at approximately 2-8 years, and then a gradual decline to their lowest densities at the oldest sites sampled (i.e. 30-31 years post-fire). This trend of post-fire reduction, recovery, and decline has also been reported in other studies. For example, Jordan (1987a) observed Southern Emu-wrens populations did not increase rapidly until a year after fire and were most abundant 2-3 years post-fire (~ 40 birds 10 ha^{-1}). McFarland (1988b) reported densities of 0.8 birds 10 ha^{-1} at 2.5 years post-fire, 1.2 birds 10 ha^{-1} at 5.5 years, and 0.4 birds 10 ha^{-1} at 6.5 years, while none were reported in the 0 and 10.5 year old heathland sites in southeastern Queensland. McFarland (1994) also reported that their highest densities of 2.0-2.5 birds 10 ha^{-1} were at 6-8 year old sites, and that no nests were found in heathlands < 2 and > 10.5 years post-fire. However, trends from these mainland studies are in contrast to those at Lake Pedder, where Southern Emu-wrens significantly increased in densities across the chronosequence (i.e. > 50 years post-fire). This may be attributed to the less productive soils and associated slower post-fire recovery of Tasmanian blanket moorlands when compared to some sites on the mainland. Nevertheless, it is not possible to determine whether a similar trend exists in other portions of the Southern Emu-wren's range since previous studies on the mainland either included a limited range of fire ages (i.e. ≤ 9 years post-fire; McFarland 1994) and/or did not provide site age data for pre-fire surveys (Loyn 1997).

Only limited and mostly qualitative information is available on the fire ecology of Striated Fieldwrens. Early records indicated the loss of natural habitat through 'burning off' (Legge 1908). Gellie (1980) stated that they are capable of finding sufficient food and cover almost immediately after fire, and have been observed foraging on recently burnt moorlands in

Tasmania. He further stated that breeding cycles may be interrupted if fires occur during spring, and breeding may not be successful for 5-7 years post-fire unless adequate unburnt patches are available; however, he provided no supporting data. Taylor *et al.*'s (1997) study on changes in the avifauna of dry sclerophyll forests with sedgy and heathy understories in southeast Tasmania indicated that the Striated Fieldwren occurred in significantly higher densities in young regrowth (0-6 years old) when compared to older regrowth coupes (6-12 years old) following clearfall logging, regeneration burns, and portions of both coupes being subsequently burnt by a wildfire; however, they were absent from mature forest (> 12 years old). Observations of Striated Fieldwrens in southwestern Australia suggest that they recolonise heath once sufficient post-fire cover is attained (Smith 1985; Hopkins and Smith 1996 in Woinarski 1999a, 1999b). Since they are considered to be an early successional species in some portions of their range, it has been postulated that long-term fire exclusion may render some heaths unsuitable for Striated Fieldwrens (Smith 1985, 1987; Woinarski 1999a). In support of this, results from Lake St Clair indicated that mean density peaked within 2-3 years post-fire at Lake St Clair and overall, there was a significant decrease in density across the full chronosequence when tested within a linear model. However, results from Lake Pedder, although not statistically significant, indicated that longer inter-fire intervals may be required for the Striated Fieldwren to reach maximum densities in lower productivity, blanket moorlands in southwestern Tasmania, since the highest densities were recorded in older sites (i.e. 31-54 years post-fire).

The only prior research focused on the Ground Parrot in Tasmania is that of Bryant (1991, 1992, 1994). Despite the fact that similar methods were used to survey Ground Parrots in this study and Bryant's, there are a number of issues that make it difficult to directly compare results. In contrast to this study, Bryant excluded sites with zero density in her analyses, and conducted global analyses across all statewide surveys and hence different locations, moorland types, productivity levels, and altitudes (Bryant 1991). She also did not differentiate between sites > 30 years post-fire for her fire age analysis, and did not clearly delineate all site boundaries by fire age (Bryant 1991, S. Bryant pers. comm. 2003). The latter may have resulted in density estimates being based upon observations of Ground Parrots that occurred across areas of multiple fire ages. Such potential sources of variation may have been exacerbated by the use of multiple observers who were probably less familiar with the boundaries of sites within a given purported fire age.

Despite the issues associated with comparing the results of this study with Bryant's, there are still some important assessments that can be made. Where possible, results from this study were re-analyzed (not reported in Results) using Bryant's method (1991) to make findings more comparable. She noted that recolonisation occurred at most sites approximately 1 year

post-fire (Bryant 1991, 1994), while results from this study indicated that Ground Parrots did not occupy sites until at least 3 years post-fire at Lake St Clair and 22 years post-fire at Lake Pedder. However, Ground Parrots have been noted to use sites soon after fires in Tasmania, in conjunction with adjacent unburnt habitat (Gellie 1980). Such younger sites, especially with small copses of older and denser vegetation, may provide adequate cover and become temporarily crowded within larger burnt areas (Meredith *et al.* 1984; Bryant 1991). Bryant (1991) found a significant positive correlation between Ground Parrot densities and fire age. When results for this study were reanalyzed, there was likewise a significant positive correlation between density and fire age across locations. Bryant reported that densities typically remained low until 4-7 years post-fire, when they peaked at approximately 2.2 birds 10 ha^{-1} . Densities were then maintained above the global mean of 1.7 birds 10 ha^{-1} (compared to a mean 1.5 birds 10 ha^{-1} for this study, excluding sites with zero density) until approximately 13 years post-fire, after which densities declined but remained above approximately 1.0 birds 10 ha^{-1} at sites 30 years post-fire and older. While the results from Lake St Clair likewise indicated that peak densities (excluding zeros) occurred at sites approximately 5 years post-fire and were then followed by a gradual decline, there was no significant relationship between density and fire age (both with and without sites with zero density). Similar to Bryant's global analyses, there was a significant positive relationship between density and fire age at Lake Pedder (both with and without sites with zero density). However, peak mean densities occurred at sites over 50 years post-fire and exceeded those of Bryant's (i.e. > 2.2 birds 10 ha^{-1}), though they were similar to the range of maximum densities that she recorded across locations (i.e. ~ 5.3 birds 10 ha^{-1}). Thus, this study indicated that the relationships between Ground Parrot densities and fire age differed notably between locations, namely low productivity, low altitude blanket moorland sites at Lake Pedder and moderate productivity, moderate altitude eastern moorland sites at Lake St Clair.

Results from the majority of studies conducted on the mainland likewise show strong associations between Ground Parrot densities and fire age and appear to reflect both the patterns of post-fire changes in densities identified in this study. These include post-fire reductions and gradual increases in densities with populations remaining present in long-unburnt sites, such as in the graminoid heathlands of Lake Pedder, and post-fire reductions followed by densities peaking and then declining toward local extinction at various ages post-fire, such as in the sedgelands of Lake St Clair. As an example of a similar pattern to that at Lake Pedder, research conducted in a heathland site at Barren Grounds Nature Reserve, New South Wales indicated that the population peaked (~ 4.0 birds 10 ha^{-1}) at 13 years post-fire (Baker and Whelan 1994, Whelan and Baker 1999) and ongoing monitoring has indicated that they are still present, although unlike at Lake Pedder apparently declining after more than 20 years post-fire (Baker 2002; Tasker and Baker 2005). Although the

Western Ground Parrot of Western Australia is a different subspecies (*P. w. flaviventris*), as at Lake Pedder they have been reported to reach their highest densities in long-unburnt heathlands (i.e. > 40 years post-fire) (Burbidge 1996, 2003; Burbidge *et al.* 2007). They can also utilise recently burnt sites (i.e. < 6 years post-fire) if they are directly adjoined by unburnt habitat with established populations, and they appear to prefer areas with a mosaic of vegetation ages (Burbidge 1996, 2003; Burbidge *et al.* 2007; Gibson *et al.* 2007). As an example of a similar pattern to that at Lake St Clair, studies conducted in Victoria showed that Ground Parrots recolonised graminoid heathlands within 3 years post-fire, gradually increased to maximum density at 15 years post-fire (i.e. ~ 3.0 birds 10 ha⁻¹), and then rapidly declined to zero at sites over 20 years post-fire (Meredith and Isles 1980; Meredith *et al.* 1984). A similar, although shortened pattern also occurred in Queensland, where Ground Parrots returned to subtropical heathland sites within 1 year post-fire, peaked at 5-8 years post-fire (4.0-5.0 10 ha⁻¹), while none were recorded at sites more than 14 years post-fire (McFarland 1991c). These examples illustrate the array of patterns of post-fire changes in densities that Ground Parrots exhibit across their range, possibly reflecting regional differences in rainfall and other factors, which in turn affect the rate of vegetation regrowth. In this sense, vegetation structure (e.g. cover) and floristics (e.g. seed production) may have a greater influence on Ground Parrot densities than fire age *per se* (Meredith *et al.* 1984; McFarland 1991c; Baker and Whelan 1994). This highlights the need for site-specific data on which to base appropriate management recommendations, particularly in relation to inter-fire intervals that will maximise likelihood of persistence over both space and time.

Overall, there were no consistent patterns in the densities of the non-resident group in relation to time since fire at either Lake Pedder or Lake St Clair. However, the necessity of pooling the non-resident species into a single group masked some species-specific patterns of density in relation to fire age. In general, the use of moorlands by the non-resident species appeared to reflect species-specific habitat and food requirements in relation to characteristics of moorlands at different seral stages (see Chapter 1). Based upon species occurrence, densities, and contributions toward community similarities at Lake Pedder and Lake St Clair, the Tasmanian Thornbill, Superb Fairy-wren, and Crescent Honeyeater dominated the non-resident group across the chronosequences. The limited information available from previous studies on the fire ecology of the non-resident species found in moorlands and adjacent vegetation communities was primarily focused on the short-term effects of fire on habitat use in comparison to 'unburnt' habitat; additional details on these effects are provided in Chapter 6. Survey data from this study indicated that there were five relatively common non-resident species found in recently burnt sites (i.e. 1-3 years post-fire) at both locations. These include Richard's Pipit, Dusky Robin, Black Currawong, Superb Fairy-wren, and Tasmanian Thornbill, although the latter two were more often recorded

utilising edge habitat (see Chapter 5). These five non-resident species primarily accounted for the higher mean densities of non-resident species when compared to those of the resident species at the youngest site surveyed at Lake St Clair (i.e. 1 year post-fire). Although the use of recently burnt habitat by these species is supported by previous observations in Tasmania and on the mainland, none appear to be restricted to early successional habitats (Green and Mollison 1961; Ratkowsky 1978, 1985; Gellie 1980; Smith 1985; Brown *et al.* 1993; Loyn 1997; Woinarski and Recher 1997; Woinarski 1999b; Paton *et al.* 2005; Recher 2005). The Crescent, New Holland and Yellow-throated Honeyeaters recorded in this study appeared to prefer mature to old-growth moorlands, consistent with studies conducted in other habitats in Tasmania (Ratkowsky 1979, 1985; Taylor *et al.* 1997). In contrast, results presented in Chapter 6 demonstrated that all three species utilised recently burnt moorlands and that short-term densities may increase under some conditions. Finally, although the Green Rosella was relatively uncommon, previous research also indicates that it more typically occurs in 'unburnt' forested habitats (Ratkowsky 1979, 1985).

Avifaunal habitat associations

Fire and its short- to long-term effects on buttongrass moorlands clearly play an important role in the varying levels of use by the resident and, to a lesser degree, non-resident bird species and subsequent patterns in densities across the fire chronosequences. However, a number of the other possible explanatory environmental variables included in this study were also found to both independently and jointly influence densities of the avifauna across the chronosequences. These variables help to further explain the apparently patchy distribution and variable densities of the avifauna, particularly the cover-dependent residents, both between sites within the same fire age class and within different fire age classes at each location.

Change in vegetation cover is the only environmental variable included in this study that is significantly related to time since fire from both a statistical and ecological perspective (see Chapters 1 and 3). The rate of vegetative recovery is determined by the respective peat soil properties and hence productivity of the sites at Lake Pedder (i.e. low) and Lake St Clair (i.e. moderate) (see Chapter 2). All three resident species are dependent on adequate vegetative cover (Meredith *et al.* 1984; Gosper and Baker 1997; Wilson and Paton 2004; see Chapter 1); however, change in cover across the chronosequences was only a significant positive factor in relation to the densities of the Southern Emu-wren and Ground Parrot at Lake Pedder. The delay in recolonisation by Southern Emu-wrens may be partly attributed to their need for adequate vegetative cover for foraging and shelter (Gellie 1980; Schodde 1982; Pickett 2000; Wilson and Paton 2004; Maguire 2006a; MacHunter *et al.* 2009). In the case of the Ground Parrot, Bryant (1991, 1994) did not find a significant correlation between

densities and total percentage cover, although she noted that recolonisation occurred at most sites approximately 1 year post-fire, when the estimated minimal projective foliage cover of 30% was typically attained. This apparent discrepancy with the results from this study at Lake Pedder may be due to the fact that she measured foliage projective cover (after Specht 1981) as opposed to using an index of vegetation cover (after Driessen 1999; see Chapter 2) and conducted her analyses across all moorland types and locations. As for the Striated Fieldwren, the Ground Parrot has been noted to be less cover-dependent than the Southern Emu-wren (Gellie 1980), presumably if adequate unburnt or partially burnt vegetation is available at the territory scale. In contrast to the low productivity sites at Lake Pedder, cover was not a significant influence on the resident or non-resident species at the moderate productivity sites at Lake St Clair. This may be attributed to the relatively rapid rate of post-fire vegetative recovery of these sites (i.e. ~ 20 vs. 5 years post-fire, respectively; see Chapter 3). From a theoretical standpoint, these observations are consistent with Fox's (1982, 1990a, 1990b) habitat accommodation model for animal succession. Fox's model states that within some faunal communities species will appear, peak, and then may disappear in a sequence that can be predicted based upon species-specific habitat requirements (e.g. low vs. high cover) and the rate of vegetative recovery. The attributes of a given fire regime (e.g. intensity, season, frequency, patchiness) may influence these successional processes and patterns in both space and time. However, it is the continuum of vegetational changes and associated habitat suitability thresholds or ranges that determines the timing of faunal succession and not time since fire *per se* (Fox 1982, 1990a, 1990b; Meredith *et al.* 1984; Baker and Whelan 1994; Monamy and Fox 2000). The differing results between avifaunal responses at low productivity sites at Lake Pedder versus moderate productivity sites at Lake St Clair and associated differences in post-fire vegetation succession provide additional support for the habitat accommodation model (see Chapter 7).

Variations in vegetation cover are not only exhibited between sites of different fire ages and locations, but also within sites due to finer-scale differences in both abiotic and biotic factors that influence patterns in vegetation composition and structure, fire effects, and post-fire recovery of buttongrass moorlands (Jarman *et al.* 1988a; see Chapter 1). The vegetation of riparian zones along the numerous watercourses that intersect the moorland sites differ in both composition and structure from the surrounding matrix and thus provide a relatively discrete avian microhabitat that is dominated by emergent shrubs, particularly in the eastern moorlands of Lake St Clair (see Chapters 1, 3, and 5). There was a significant, positive relationship between mean densities of the Southern Emu-wren and the areal percentage of riparian zones at Lake St Clair, which accounted for the highest independent contribution of any variable for the Southern Emu-wren at either location. There were also positive, although non-significant, relationships between mean densities of the Southern Emu-wren

and riparian area at Lake Pedder, as well as the Striated Fieldwren at both locations. The higher fuel moisture levels associated with small drainages can limit or prevent combustion with the result that unburnt or partially burnt riparian zones serve as 'biological legacies' (*sensu* Franklin *et al.* 2000) that both directly and indirectly contribute to post-fire recovery. For example, in the case of the one recently burnt site at Lake St Clair (HAR, ≤ 1 year post-fire), an area search the day after the fire (T. Chaudhry unpublished data) as well as data from the distance surveys indicated that the unburnt riparian vegetation along the drainages provided adequate cover to enable Southern Emu-wrens to at least periodically use the site and enable Striated Fieldwrens to occupy it continuously post-fire. These results are consistent with previous research on the Australian mainland and abroad that likewise indicated that such small unburnt or partially burnt patches of vegetation in sedgeland and heathland habitats, particularly if available at the territory scale, may enable Southern Emu-wrens and other cover-dependent, territorial passerines to persist in the post-fire environment, although often in reduced numbers (Recher 1981; Recher and Christensen 1981; Rowley and Brooker 1987; Reilly 1991a, 1991b; Beyers and Wirtz 1995; Pickett 2005; Bain *et al.* 2008).

Furthermore, even in highly fragmented habitats such as those found in the Mt Lofty Ranges of South Australia, observations of Southern Emu-wrens indicate that they are capable of persisting and possibly breeding in long and narrow or very small patches of high quality swamp vegetation (Littely and Cutten 1996). Although Gellie's (1980) observations of Striated Fieldwrens in Tasmanian moorlands indicated that they can persist in recently burnt moorlands, observations from this study suggested that such use may depend on the availability of adequately sized and dispersed patches of unburnt or partially burnt post-fire refugia that are often associated with riparian zones. In contrast to the Southern Emu-wren and Striated Fieldwren, relationships between mean densities of the Ground Parrot and the non-resident species with riparian area differed between locations. Along with fire age, riparian area maximised the rank correlation between similarity matrices and avifaunal composition and densities at Lake St Clair, indicating that both variables help to account for, although they are not necessarily causal of, inter-site similarities at the community level (Clarke and Gorley 2001). Overall differences in results between locations at the species and community levels may be attributed to the greater structural and floristic diversity across fire ages provided by riparian habitats at Lake St Clair compared to the sedge-dominated matrix of these eastern moorlands, whereas such differences between habitats are less pronounced in the scrub-dominated blanket moorlands at Lake Pedder (see Chapter 3).

Results from Chapter 5 indicated that the use, and hence value, of riparian zones for the resident and non-resident bird species in both low and moderate productivity moorlands was

disproportionately high compared to their extent in the moorland landscape (i.e. < 5% of study area). Both the Southern Emu-wren and Striated Fieldwren used riparian habitats significantly more than expected across the full range of fire ages at both locations and both the resident and non-resident groups were more likely to select riparian habitat at Lake St Clair when compared to Lake Pedder. In the case of moorlands of moderate productivity at Lake St Clair, where the arthropod surveys were conducted, selection for these habitats by insectivorous birds reflected the greater availability of potential arthropod prey across fire ages. Furthermore, peak mean densities of the Southern Emu-wren and Striated Fieldwren overlapped with those of mean abundance and energy content of arthropod prey resources in relation to fire age. However, the discrepancy between years of sampling and fire ages between the avian and arthropod surveys made it difficult to provide any definitive statements regarding their relationship and possible interaction with fire. Nevertheless, the evidence suggested that sites with higher proportions of riparian zones, at least at Lake St Clair, provide preferred riparian microhabitats that support higher carrying capacities and thus densities of the two insectivorous resident species. At least in the case of the Southern Emu-wren, this was associated with significantly higher densities at sites with greater proportions of riparian areas. Possible mechanisms underlying the higher relative selection probabilities for riparian zones in relation to matrix habitat are detailed in Chapter 5.

The spatial configuration of vegetation communities at the site to landscape scales were also related to avifauna densities to varying degrees. Site area was a significant negative factor for the Southern Emu-wren and Ground Parrot at Lake Pedder and non-resident species at Lake St Clair, while it was a significant positive factor for the Ground Parrot at Lake St Clair and in relation to inter-site similarities at the community level at Lake Pedder. However, elucidating the effects of site area on densities is complicated by a number of factors, including criteria used for the selection and delineation of sites (see Chapter 2), habitat use (e.g. higher selection of edge habitats by non-resident species; see Chapter 5), the relatively well-connected moorland matrix, especially at Lake Pedder, and the fact that site area is a component of, and significantly correlated with, other spatial metrics (e.g. shape index) (see Chapters 2-3). In this sense, site area as defined may not be commensurate with the characteristics that influence individual and species-specific site selection (i.e. from an 'organism's perspective' *after* McGarigal and Marks 1995) and hence densities. Accordingly, the underlying mechanisms that contributed to the observed patterns are unclear, but may be partly influenced by the independent and joint influence of other habitat variables. Such locational variability in the influence of patch area on bird species abundance and richness has been found in other studies and likewise attributed to scalar issues and the influence of other location-specific habitat variables (e.g. McGarigal and McComb 1995; Mac Nally and Watson 1997; Johnson and Igl 2001). In relation to the

Ground Parrot, it should be noted that although area was relatively weakly related to densities at both locations, the logistic analysis indicated that area did not have a significant influence on occupancy in either case. This may be attributed to the minimum moorland patch size (i.e. 5 ha) used as a site selection criterion to ensure that sites were of an adequate size to support at least one breeding pair of the three resident species (see Chapters 1-2). In relation to edge contrast index and the percentage of suitable habitat, results indicated that Ground Parrot occurrence and densities were significantly higher at sites with lower contrast edges at both locations, and at sites at Lake St Clair with a higher percentage of suitable habitat within a 400 m radius of survey points. There were weaker, although still positive relationships between occurrence and densities with the percentage of suitable habitat at Lake Pedder. On average, sites at Lake Pedder had notably lower edge contrast and a higher percentage of adjacent suitable habitat, reflecting the characteristic ecological differences and hence distribution of blanket versus eastern moorlands within the landscape mosaic (see Chapter 1). Nevertheless, at either location Ground Parrots did not occur at sites with < 45% of suitable habitat within the 50 ha circles, indicating a threshold response within the study area. A study conducted in subtropical heathlands in New South Wales likewise indicated that Ground Parrots required relatively large tracts of suitable habitat (i.e. > 300 ha) to enable occupancy (Martin and Catterall 2001), with the higher threshold likely due to the highly fragmented condition of heathlands on the mainland when compared to the relatively extensive and contiguous tracts of suitable habitat found in the TWWHA (see Chapter 1). Although Ground Parrots are relatively strong fliers, have been recorded dispersing over long distances, and have been noted dispersing between suitable habitat patches separated by non-habitat (Forshaw 1981; see Chapter 1), sites that are primarily bordered by high contrast woodlands and forests and isolated from other moorlands possibly inhibit dispersal and persistence, at least in the short-term. These relationships between Ground Parrot occurrence and densities and the spatial configuration of moorlands may help to partially explain the relatively patchy distribution of Ground Parrots throughout the study area, and particularly at Lake St Clair, when compared to the widespread Southern Emu-wren and Striated Fieldwren.

Limitations

The results from this study highlight the complexity of fire-adapted ecosystems and inherent difficulties in elucidating patterns in avifaunal composition and densities and their underlying processes. The limitations of space-for-time (SFT) study designs in particular, and fire ecology studies in general, as well as the measures incorporated into this study to help ameliorate them are detailed in Chapter 2 and herein. The primary limitation of SFT studies is that inferences may be weakened due to the possibility of inter-site variability in abiotic and biotic factors confounding or compounding observed differences in populations

from any effects of fire *per se* (McFarland 1991c; Whelan 1995; Loyn 1999; Woinarski 1999b; Paton *et al.* 2005). Previous research on Australian heathland birds indicates that populations may fluctuate greatly from year to year due to climatic and other factors, highlighting the limitations of short-term studies such as this one (i.e. only 1 year of observations), while some sites may support higher species richness irrespective of fire age (Rowley and Brooker 1987; Maron *et al.* 2005; Paton *et al.* 2005). In addition, apparent changes in species composition and densities after a given fire event may not be representative of 'typical' responses due to differences in fire-specific and regime-specific characteristics (Loyn 1999; Woinarski 1999b; Bradstock *et al.* 2005; Bain *et al.* 2008; see Chapter 2). Furthermore, there may be time lags between a fire event (or events) and faunal responses at the individual and population levels; thus, observed patterns may reflect conditions of a younger age class as well as longer-term site history (Wiens 1985; Knick and Rotenberry 2000). This wide range of potential explanatory spatiotemporal variables cannot typically be measured and included in a given study due to resource constraints and analytical limitations such as those resulting from highly stratified study designs and limited sample sizes, as was the case for this study (Buckland *et al.* 2001; Marques *et al.* 2007). Such limitations may also increase bias and the possibility of both Type I and Type II errors, but can be partially addressed by the *a priori* incorporation of applicable theory and adequate replication, as was attempted for this study (Whelan 1995; Anderson *et al.* 2001a, 2001b; Buckland *et al.* 2001; Quinn and Keough 2002).

The results from this study are further discussed in Chapter 7, particularly in relation to the persistence of the resident species through changes in fire regimes since European colonisation despite exhibiting fire-sensitive attributes, and their overall implications for avifaunal conservation and fire management of Tasmanian buttongrass moorlands.

Avifaunal habitat use and potential availability of arthropod prey resources in relation to post-fire succession of buttongrass moorlands in the Tasmanian Wilderness World Heritage Area

Introduction

Avian ecology has long been dominated by habitat use and selection studies (for reviews see Hildén 1965; Block and Brennan 1993; Jones 2001), as the relative mobility and ubiquity of birds enable them to choose among habitats more readily than most other taxa (Cody 1985). Habitat selection is a process carried out by individuals that may reflect a demonstration of choice (i.e. preference) between available habitat types, while habitat use is the resulting pattern of the distribution of those individuals across habitat types (Hutto 1985; Jones 2001). These processes and resulting patterns may manifest themselves at different scales (Johnson 1980; Wiens 1985): from geographical ranges (Lee and Rotenberry 2005), to home ranges (Luck 2002), to microhabitats within home ranges (Maguire 2006a), and finally to the procurement of resources such as food items within microhabitats (Johnson 2000). To determine actual preference for a particular habitat, it is necessary to quantify both the relative usage (i.e. quantity utilised) and spatiotemporal availability (i.e. accessibility) of habitats at the relevant scales of interest (*sensu* Johnson 1980). The underlying assumption is that habitat use is selective if it is disproportionate to availability, and that demonstrated preferences are adaptive in that they confer fitness advantages to those individuals; although this assumption is seldom explicitly assessed by most researchers (Johnson 1980; Garshelis 2000; Jones 2001).

Avian habitat relationships are influenced by a wide range of biotic and abiotic factors, such as vegetation structure (MacArthur 1958; Gilmore 1985), floristics (Rotenberry 1985; Mac Nally 1990), food resources (Karr and Brawn 1990; Brodmann *et al.* 1997), soil types (Ormerod *et al.* 1991), and disturbance such as fire (Baker 2000; Ward and Paton 2004a, 2004b). At the home range scale, intrinsic factors such as vegetation characteristics and the relative availability of suitable food resources are considered to be primary factors influencing the allocation of time and energy by individuals and the resulting patterns of microhabitat use (Hutto 1985; Morris 1987; Block and Brennan 1993; Maguire 2006a, 2006b). In pyrogenic landscapes, fire can alter these factors and result in significant changes in avian habitat use, distribution, and abundance (Wooller and Calver 1988; Stuart-Smith *et al.* 2002; Lindenmayer *et al.* 2009). However, developing a sound understanding of such complex interactions often requires long-term studies (Brooker 1998) or space-for-time

(SFT) substitutions when such studies are not feasible (Pickett 1989; Loyn 1999; Woinarski 1999b).

Buttongrass moorland is a globally unique and highly pyrogenic vegetation community (Brown *et al.* 1993; Balmer *et al.* 2004; Harris and Kitchener 2005) that covers extensive areas in Tasmania (0.55 million ha), and most notably in the Tasmanian Wilderness World Heritage Area (TWWHA, 335,000 ha) (Balmer *et al.* 2004; TVMP 2004; see Chapter 1). Despite such expanses, buttongrass moorlands support a relatively depauperate avifauna that is comprised of only three cryptic ground-dwelling resident species that are known to depend exclusively upon moorlands for survival and reproduction within the study area, namely the Southern Emu-wren (*Stipiturus malachurus*), Striated Fieldwren (*Calamanthus fuliginosus*), and Ground Parrot (*Pezoporus wallicus*), and a small number of transient species that are typically associated with adjacent woodland and related habitats (Brown *et al.* 1993; see Chapters 1, 4, and 6). Although buttongrass moorlands have been regarded as a structurally uniform habitat for birds (Brown *et al.* 1993), they are comprised of a range of vegetation communities that differ in both structure and floristics, particularly in response to fire (Jarman *et al.* 1988a; see Chapters 1 and 3).

To date, only Bryant (1991, 1994) has conducted a study on habitat use and the diet of a Tasmanian buttongrass moorland resident; however, this was limited to the granivorous Ground Parrot and did not consider the effects of the availability of habitat or food resources on habitat selection. The Southern Emu-wren, Striated Fieldwren, and the avian community as a whole have not been studied in Tasmanian moorlands. On the Australian mainland, only one detailed use versus resource availability study has been conducted by Maguire (2006a) on the Southern Emu-wren in Victoria; however, this was on a different subspecies, in an area with highly fragmented habitats unlike those found in the TWWHA, and did not consider the effects of fire. Other studies on the mainland such as that by McFarland (1991a) on the Ground Parrot in Queensland did not formally analyse use versus availability and that by Gosper and Baker (1997) on the Striated Fieldwren in New South Wales only provided a qualitative assessment of occupied habitat (Gosper and Baker 1997). Relatively few studies in general have focused on the food resources available to insectivorous birds (Poulin and Lefebvre 1997). Only one study by Maguire (2006b) has examined nestling provisioning and arthropod availability of the Southern Emu-wren. Even fewer have investigated the effects of fire on food resources of the Australian avifauna in any more than a speculative manner (Woinarski 1999a), and no such studies have been attempted in buttongrass moorlands. Previous research on terrestrial arthropods in buttongrass moorlands, particularly in relation to fire, has been limited to studies conducted by Greenslade (1997), Greenslade and Driessen (1999), Greenslade and Smith (1999) and

Driessen and Greenslade (2004), and a collaborative SFT research project (M. Driessen unpublished data). Clearly, there is a significant need for multidisciplinary research on the effects of fire on avian habitat use and food resources in order to provide more ecologically sound and holistic guidelines to resource managers (Woinarski 1999b).

This study compares habitat selection by the buttongrass moorland avifauna between three habitats at different stages of post-fire succession and attempts to relate that use to the potential availability of arthropod prey. The primary aims of this study were 1) to investigate whether the avifauna use habitats disproportionately in relation to availability and fire age within low productivity blanket moorlands at Lake Pedder and moderate productivity eastern moorlands at Lake St Clair in the TWWHA; 2) to investigate whether patterns of habitat use of insectivores reflect the potential availability of arthropod prey resources in relation to habitat type and fire age at Lake St Clair; 3) to describe and compare the terrestrial arthropod community between moorland matrix and riparian habitats; and 4) to provide information to assist with fire management and conservation of the buttongrass moorland avifauna in the TWWHA.

Methods

General rationale

Habitat use data were collected during the course of distance sampling surveys conducted throughout 2004 for the space-for-time study presented in Chapter 4. Since distance sampling was primarily intended to collect data on bird densities, behavioural observations were limited to the initial visual observation to provide a general sense of the behavioural cues eliciting an observation. After the completion of the 2004 SFT surveys, a pilot study was conducted to assess the feasibility of conducting detailed behavioural surveys, with the primary goal of obtaining data on foraging behaviour for the Southern Emu-wren and Striated Fieldwren. Such quantitative measures of foraging behaviour have been recommended by Hutto (1990) to provide a way of checking whether measures of food availability are biologically meaningful. However, it was determined that such a study was not feasible in consideration of the limited time and resources remaining for this thesis. Furthermore, it should be noted that such comprehensive studies on the Southern Emu-wren and Striated Fieldwren are inherently difficult due to their cryptic nature, in addition to the other issues associated with obtaining detailed behavioural observations in the field. This difficulty has been noted by other researchers (M. Pickett pers. comm. 2005; J. Baker pers. comm. 2005; Maguire and Mulder 2004) and reflected in the lack of formal studies on foraging behaviour in the literature (for reviews see Higgins *et al.* 2001; Higgins and Peter 2002), with the recent exception of Maguire (2006b). In the absence of adequate data on the foraging behaviour of the target species in Australia in general, and Tasmanian moorlands in

particular, it is assumed that foraging time spent in the specified habitats is roughly proportional to overall habitat use and is thus indicative of the relative importance of these habitats to their feeding ecology (Morse 1971; Hutto 1985).

An initial assessment of the habitat use data indicated that the avifauna appeared to be selecting both riparian and edge habitats disproportionately to their availability within both study locations and that the strongest preference seemed to be for riparian habitats at Lake St Clair. To investigate some of the potential factors that may have been contributing to these observed patterns, a *post hoc* study of the availability of food resources was initiated but was limited to comparing only matrix and riparian habitats at Lake St Clair due to resource and time constraints. In addition, as over 95% of habitat use observations were of species that are either entirely or partially insectivorous (see Chapter 1), the investigation of food resources was limited to potential arthropod prey. Although it would have been desirable to conduct this study synchronously with the habitat use surveys, the arthropod sampling could not be conducted until the summer of 2006. This delay limits the strength of inferences that can be drawn between the habitat use and prey resource data largely due to associated post-fire successional changes in vegetation and, in turn, those of the arthropod and avian communities. Furthermore, two of the study sites were burnt by PWS during 2005. Therefore, site ages have been changed accordingly between the two datasets. Despite these obvious limitations, these studies are presented together to preserve the original intent, but the two datasets are not formally analysed together in recognition of these issues. Finally, for simplicity the term ‘habitat’ is used throughout this paper, although according to some definitions the vegetation communities identified in this study (i.e. matrix, riparian zone, edge) can also be regarded as ‘microhabitats’ as they are the primary habitat components within individual home ranges in the study area (Morris 1987; Block and Brennan 1993).

Study area

Details on the study design and area, and site descriptions, fire histories, vegetation configurations, and floristics are provided in Chapters 2 and 3. Analyses were conducted primarily in relation to site attributes including proportional habitat area (i.e. matrix, riparian zone, edge) and fire age (years post-fire). The habitat use study included 10 of the 12 sites at Lake Pedder (3-54 years post-fire) and 12 of the 14 sites at Lake St Clair (1-31 years post-fire) primarily within the TWWHA as well as adjacent land allocations. The four sites excluded from the analyses (i.e. MCR, CCN, SCN, GIT) did not contain all three habitat types and their inclusion would have violated one of the underlying assumptions of the habitat use analyses: all habitat types must be accessible to all birds at the appropriate scale (Manly *et al.* 2002). The arthropod availability study was conducted at 11 sites at Lake St Clair, also excluding SCS, as PWS conducted prescribed burns at the site during autumn

2005 and again in spring 2005 because the first burn failed to meet fuel reduction objectives (T. Norris pers. comm. 2005). In addition, prescribed burns were conducted at TRR on April 21, 2005 and NPE on September 21, 2005 (see Chapter 6). Thus, these sites were reclassified as (less than) one year old, while the remaining sites were reclassified as being two years older than as defined in Chapter 2 and were analysed as exact ages (i.e. < 1-32 years post-fire).

Habitat use surveys

Habitat use data were collected through the course of distance sampling surveys conducted during the summer (2 February - 9 March), winter (24 May - 6 July), and spring (11 October - 27 November) of 2004. Details on the study design, fire histories, and survey methods and conditions are provided in Chapters 2- 4. Only initial visual observations of birds utilising habitats within the sites were recorded to determine proportional usage, as recommended by Bell *et al.* (1990). This was also done to eliminate the potentially high bias if auditory cues alone were used to estimate which habitats were being used. Birds observed flying were only recorded if their subsequent movements indicated that they were using specific habitat features on the ground. Thus, aerial feeders such as Welcome Swallows and other birds that were observed flying above or across the site were excluded from consideration. In addition, since mature trees in adjacent woodland edges and large wood copses were not considered to be affected by the moorland surface fires, birds observed in the primary canopy were likewise not recorded. Care was taken to avoid double counting of individual birds by noting the observation time, their relative location (e.g. left or right of transect), and maintaining a mental map of their locations. Conspecifics that were observed in close proximity to each other (i.e. $\sim < 10$ m apart) and exhibited behaviours consistent with those of a pair bond or family group were recorded as a cluster. All such visual observations were recorded as occurring in either the matrix, riparian, or edge habitats along with any details of the habitats being used (e.g. structure and location).

Species-specific habitat use could not be analysed due to limited sample sizes (see below); therefore, all species recorded were classified into two groups based on their known habitat associations (see Chapter 1). The resident group includes the three species that exclusively depend upon and breed within the moorlands in the study area, namely the Striated Fieldwren, Southern Emu-wren, and Ground Parrot (Brown *et al.* 1993). The non-resident group is comprised of the remaining 18 species that are only marginal users of moorlands and are all typically associated with and breed within adjacent woodland and other non-moorland habitats. Non-residents of moorlands include: Beautiful Firetail, Black Currawong, Black-faced Cuckoo-shrike, Black-headed Honeyeater, Crescent Honeyeater, Dusky Robin, Flame Robin, Green Rosella, Grey Fantail, Grey Shrike-thrush, New Holland

Honeyeater, Pink Robin, Richard's Pipit, Strong-billed Honeyeater, Superb Fairy-wren, Tasmanian Thornbill, Yellow Wattlebird, and Yellow-throated Honeyeater (see Chapters 1 and 4).

Habitat use modelling

Habitat use versus availability studies are common throughout the wildlife literature and a range of statistical techniques have been developed and debated over the years (e.g. Johnson 1980; Alldredge and Ratti 1986; Thomas and Taylor 1990; Alldredge and Ratti 1992; Aebischer *et al.* 1993; Alldredge *et al.* 1998; Garshelis 2000; Jones 2001; Manly *et al.* 2002). The underlying philosophy behind these studies is that use of different habitats is considered to be selective if it is disproportionate to their availability, which is typically assumed to be the proportional areas of habitats within a study area (Johnson 1980; Alldredge *et al.* 1998). Following Manly *et al.* (2002), this study utilises a common sampling design and protocol in which the available habitat types are mapped, used habitats are randomly sampled from the populations in the study area, and individual animals are not uniquely identified. They highlighted a number of key assumptions that are necessary for the valid application of resource selection analyses that are outlined below as they pertain to this study:

- Bird habitat use was randomly sampled from the distance transects as described in Chapter 4.
- Clusters (pairs or family groups) of birds, when present, were treated as independent units of observation (Neu *et al.* 1974).
- Observations were recorded during three seasons to encompass seasonal variability and separated by at least three months such that relocations of the same individuals or clusters were assumed to be independent of each other.
- All habitats were assumed to be available to all birds at the site scale (i.e. within whole or partial home ranges). However, it was necessary to pool habitat types within fire age classes and locations in order to meet sample size requirements (see below).
- Availability of habitats were known and derived from the fine-scale GIS habitat map described in Chapter 3 and estimated locations of habitat use observations have been cross-checked with the map to ensure they were classified correctly.
- Availability was constant as the boundaries between the moorland matrix and adjacent habitat types were known to be stable over the study period (pers. obs.) and intra- and inter-specific habitat selectivities were assumed to be constant over time.
- Birds were assumed to be detected with an equal probability in all the habitats. Although detectability is addressed in Chapter 4 in relation to distance sampling density estimates, Manly (2002) pointed out that when transects are used to collect

count data on used resource units the probability of detection may similarly depend both on the distance from the line and the type of habitat. He proposed some logistic regression models that would incorporate these potential confounding factors; however, as these methods are still in development and require moderate sample sizes they could not be readily applied to these data.

In consideration of the study design, validity of the above assumptions, and the nature of the study questions, log-linear modelling was chosen to examine relative selection of the matrix, riparian, and edge habitats in relation to fire age. As originally proposed by Heisey (1985) and more recently described by Manly *et al.* (2002), log-linear modelling is used to develop a resource selection function ($w(x)$) (McDonald *et al.* 1990) based on Manly's selectivity index, which is a weighting factor reflecting the non-random use of resources (Manly 1974; Heisey 1985). The estimated value for a resource unit is proportional to the probability of its use (Manly *et al.* 2002), thus resource selection functions provide a unified theoretical framework for such resource selection studies (Heisey 1985; Alldredge *et al.* 1998).

A hierarchical log-linear model building process was used to determine the models of best fit from which to calculate the resource selection function and relative selection probabilities (Heisey 1985; Manly *et al.* 2002). Consistent with the design of this study, habitat use data at Lake Pedder and Lake St Clair were analysed separately (see Chapter 2). The models were programmed in R 2.0.1 (R Development Core Team 2006) with a log link function, Poisson error distribution, and with base rates (i.e. offset terms) set as the proportional areas of each habitat within the study area (Manly 1992), thus reflecting their differential availabilities (Manly *et al.* 2002). Initial examination of residuals and Akaike Information Criterion (AIC) values indicated that the limited sample size ($n = 108$ at Lake Pedder, $n = 142$ at Lake St Clair) and inclusion of all the factors (i.e. season, fire age, site, habitat, and species) within a full model for each location did not provide adequate power or acceptable error rates, as most of the cells did not have expected values > 5 and the total samples size was not five times the number of cells in the table (Manly *et al.* 2002). Part of this complication also resulted from the inherently small proportions and hence small expected counts in riparian and edge habitats since each comprised $< 10\%$ of total area at the site scale while the matrix comprised $> 85\%$. However, pooling these habitats further did not make any ecological sense and the issue of dealing with habitat classes that comprise such small relative proportions is not otherwise dealt with in the literature. As a result of these problems, separate two way models had to be developed to investigate the primary research questions of interest at both study locations, namely if there was selection of habitat by group across fire age classes and across the residents and non-residents within each fire age class. Despite the fact that it is often necessary and justifiable to pool and limit factors to

meet statistical considerations in avian ecology studies (Noon and Block 1990), it is recognised that some of the other possible explanatory variables that could not be included in the models (e.g. season and species) probably do influence habitat selection to some unknown degree. However, as the relative habitat proportions were fairly similar among sites, pooling area estimates across sites and fire age classes probably had little effect on the estimated habitat availabilities and overall results. Given such assumptions and limitations, inferences are necessarily restricted to showing relative selection of the defined habitats and populations (Thomas and Taylor 1990; Alldredge *et al.* 1998). Analysis of deviance tables were calculated for these models based on the likelihood ratio chi-square goodness-of-fit statistics and the models of best fit for each analysis were used to generate the associated parameter estimates (Heisey 1985). The parameter estimates were then used to estimate the resource selection functions and relative selection probabilities for each habitat based on a standardised unit of 1.00 for the matrix, as described by Manly *et al.* (2002).

Arthropod prey sampling

To measure the availability of potential prey of insectivorous birds it is necessary to identify the foraging methods of the target species over available microhabitats, their potential prey taxa, and the abundance (relative or absolute) of those taxa in the relevant microhabitats (Wolda 1990; Johnson 2000). Table 1 provides a summary of the foraging methods, microhabitats, and arthropod prey orders identified from the literature and confirmed by personal observations in the study area, as well as a summary of the relevant habitat use data that are more fully described herein. Potential arthropod prey were identified from the literature based on the insectivorous species that accounted for at least 5% of total habitat use observations ($n = 250$). Information available on the diet of bird species that are found in moorlands are very limited and are primarily based on observational data that have been recorded in other vegetation types (e.g. Fletcher 1915a) and on limited stomach contents analyses conducted on Australian mainland subspecies (e.g. Barker and Vestjens 1989; for details refer to Higgins *et al.* 2001, Higgins and Peter 2002). Dietary data were only consistently available at the ordinal level and it was considered to be overly restrictive to only record the relatively few families and species for which data were reported. Therefore, identification was limited to order and life form (larvae, nymph, or adult) to match the taxonomic level available from the literature (Johnson 2000). Those orders not known to be utilised as prey were excluded from analyses; however, it is recognised that some excluded orders may serve as prey under some circumstances. Similarly, only very limited data were available regarding the size of prey items utilised by any of the insectivorous species and no minimum or maximum prey size thresholds could be determined from the literature (range < 2 - > 27 mm) (Paton 1982; Tullis *et al.* 1982; Maguire 2006b). Therefore, all arthropods

within the identified prey orders were included regardless of size, with the recognition that size preferences probably do exist among the moorland avifauna.

Table 1. Foraging ecology and habitat use of insectivorous birds in buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. Species included accounted for at least 5% of total habitat use observations ($n = 250$) that included but were not limited to foraging behaviours. Diet and foraging data are from the literature and confirmed by pers. obs. where noted* (North 1912; Fletcher 1915a; Lea and Gray 1936; Barker and Vestjens 1989; Gosper and Baker 1997; Higgins *et al.* 2001, Higgins and Peter 2002, and references therein; Maguire 2006b, see Chapter 1).

Species	Foraging methods	Foraging microhabitats	Arthropod prey orders	Habitat use in study area
Southern Emu-wren	Gleaning and sallying*	Not well known; ground and litter; foliage and inner twigs of low, dense herbs and shrubs*	Araneae, Coleoptera (incl. larv.), Diptera (incl. larv.), Hemiptera, Hymenoptera, Lepidoptera (incl. larv.), Mantodea, Neuroptera, Orthoptera*	Matrix 64%, Riparian 36% ($n = 44$)
Striated Fieldwren	Not well known, gleaning *	Not well known; ground and litter; foliage and branches of low herbs and shrubs*	Araneae, Coleoptera, Hemiptera, Hymenoptera, Lepidoptera (incl. larv.), Neuroptera, Orthoptera*	Matrix 74%, Riparian 19%, Edge 7% ($n = 84$)
Superb Fairy-wren	Gleaning and sallying*	Ground and litter; low understorey foliage and twigs of shrubs and trees*	Araneae, Chilopoda, Blattodea, Coleoptera (incl. larvae), Dermaptera, Diptera, Hemiptera, Hymenoptera, Lepidoptera (incl. larvae), Neuroptera, Orthoptera	Matrix 13%, Riparian 30%, Edge 57% ($n = 23$)
Tasmanian Thornbill	Not well known, gleaning and sallying*	Not well known; foliage and twigs of shrubs and trees*	Araneae, Coleoptera, Diptera, Hemiptera, Hymenoptera, Lepidoptera larvae	Matrix 25%, Riparian 25%, Edge 50% ($n = 20$)

Many methods are used to sample arthropods in ornithological studies. Sweep netting is probably the most commonly used as it is simple and inexpensive; however, it is often of limited value in making direct comparisons between habitats because it can only provide an index of relative abundance (Cooper and Whitmore 1990). Furthermore, the efficiency of sweeping can vary due to differences in vegetation structure (Ausden 1996), and sweep nets are not able to penetrate into the interior and lower portions of shrubs (Osborne and Allen 1999). This is compounded by daily activity patterns of arthropods that may result in biased samples depending on the time of day and height of vegetation being sampled (New 1998; Southwood and Henderson 2000). Vacuum sampling is one alternative to sweep-netting that enables the estimation of absolute abundance (or density) but it has not been broadly used in avian research to date (Cooper and Whitmore 1990). Vacuum sampling has traditionally been used in studies of arthropod pests in structurally homogenous agricultural systems (e.g. Ellington *et al.* 1984; MacLeod *et al.* 1994); however, it is increasingly being used for

ecological research in structurally complex habitats such as coastal sage scrub (Bolger *et al.* 2000), chaparral (Osborne and Allen 1999), and forest understoreys (Moir *et al.* 2005), and has been trialled in one prior study in buttongrass moorland (Greenslade 1997). Buffington and Redak (1998) compared the effectiveness of sweep-netting and vacuum sampling in coastal sage scrub in southern California and found that vacuum sampling was more efficient overall, sampled greater species richness, and collected more individuals per order (most notably for Diptera, Hemiptera, and Hymenoptera). Vacuuming can also be more effective at capturing smaller arthropods (e.g. parasitic Hymenoptera) (Buffington and Redak 1998), as well as those that inhabit the inner and lower portions of shrubs (e.g. Hemiptera) and the ground litter (e.g. Coleoptera) (Stewart and Wright 1995; Brodmann and Reyer 1999):

representing orders that are known to be consumed by all of the target bird species.

However, vacuum sampling may under-sample some arthropods, such as those that may take evasive action due to the disturbance caused by the vacuuming process (e.g. Orthoptera) or that are firmly attached to vegetation (e.g. Lepidoptera larvae) (Ausden 1996). Additionally, vacuums can also be susceptible to mechanical failures, which are not an issue with many other methods. Despite limitations inherent with any sampling method, vacuum sampling is considered to be less biased overall and has been recommended over other common sampling techniques, and sweep-netting in particular (Cooper and Whitmore 1990; Ausden 1996; Buffington and Redak 1998; Southwood and Henderson 2000; Moir *et al.* 2005).

Based on the above attributes, it was determined that vacuum sampling was the most appropriate method that would minimise biases between structurally diverse habitats (i.e. matrix and riparian zones) and collect a broad range of both active (e.g. flying insects caught by sallying) and inactive (e.g. sedentary insects caught by probing) potential prey taxa from the primary microhabitats known to be used by the target species (see Table 1).

Vacuum sampling was conducted with a 25 cc, 2-stroke gas trimmer (Model PS-06152, Ryobi) with vacuum attachment (Model RLV1100, Ryobi) and a modified suction tube constructed from a 10 x 70 cm PVC pipe. Collection bags measuring ca 17 x 37 cm were sewn from fine nylon voile (0.2 mm x 0.2 mm mesh size), and held inside the suction tube by folding the hem back over the tube opening and securing it with a snug-fitting PVC collar. Mean air intake velocities at full throttle and with an empty collection bag in place were measured at the opening of the suction tube with a hand-held anemometer (Skymate+Plus Model SM-19, Speedtech Instruments) every ten seconds over one minute. The mean velocity of two trials was 41.2 m s^{-1} , which is within the range of other vacuum sampler velocities (e.g. 20 m s^{-1} , Osborne and Allen 1999; 24.8 m s^{-1} , Buffington and Redak 1998; 45.6 m s^{-1} , Stewart and Wright 1995), and well exceeds the recommended minimal velocity of 26.8 m s^{-1} (Southwood and Henderson 2000).

Vacuum sampling was performed at each of 11 sites at Lake St Clair from 14-17 February 2006, the month that is known to have peak arthropod abundance in moorlands (M. Driessen unpublished data) and during which summer bird surveys were conducted in 2004 for the SFT study. Site sampling order was determined systematically to reduce potential biases between fire ages. Sampling was conducted during daylight hours and limited to fine weather with calm to light winds and dry vegetation (Sutherland 1996; Ozanne 2005); however, it should be noted that during sampling and most of the year the peat soils are water-logged and the sites often have standing water. The mean temperature was 23.5°C (range 19-27°C) and the mean relative humidity was 51.6% (range 38-76%) with variable cloud cover. Although these are typical conditions for the study area in February (see Chapter 2), multiple linear regression was used to explore potential sampling biases of time of day, temperature, humidity, and cloud cover on mean arthropod energy content by habitat and site (wind was excluded from the analysis as speeds were fairly consistent). No significant relationships were found ($F = 0.194$; $df = 4, 17$; $P = 0.938$, $R^2 = 0.044$) and variance-inflation factors (all < 10) did not suggest any issues with collinearity of the explanatory variables (Quinn and Keough 2002), indicating that these factors had limited influence on the overall results. These findings are consistent with those of Greenslade and Driessen (1999), which showed total epigaeic arthropod numbers collected by sweep-netting in Tasmanian buttongrass moorlands were likewise not related to time of day or temperature.

Arthropods were sampled using a split-plot design, with factors including fire age, sites (i.e. replicates) nested within fire age, and habitat (Quinn and Keough 2002). Three subsamples consisting of 1 x 1 m quadrats were taken within each matrix and riparian habitat at each site to provide a more accurate measure of food availability. This was deemed to be a reasonable scale of measurement considering that the resident Southern Emu-wren and Striated Fieldwren are known to thoroughly work their way through shrubs at this scale while foraging (see Chapter 1) and due to the anticipated large sample sizes of arthropods (based on results from a pilot study). Quadrats were located at 20 m intervals along a 40 m transect randomly positioned and oriented within each habitat, with a minimum buffer of 20 m from adjacent habitats using ArcView GIS 3.3 (ESRI Inc. 2002) and the random point generator extension (Jenness 2005). For logistical and safety reasons, transects were limited to within 250 m of the point of access of each site, or the next nearest area. A total of 66 quadrats were taken among the two habitats and 11 sites. A three-sided PVC quadrat was slid into place to minimise disturbance to the plot. All vacuuming was conducted by the author to eliminate inter-operator bias. The vacuum was started immediately prior to sampling with a plastic plug in the nozzle to prevent sampling outside of the specified time and area. The plug was then removed and vegetation within quadrats was thoroughly vacuumed by sweeping the nozzle both vertically and horizontally at full throttle for one minute from

ground level to the maximum height that the vacuum could reach (~ 2 m) and corresponding to the height within which the majority of habitat observations were recorded for the species considered herein (pers. obs.) (Figure 1). The sampling was conducted in a manner that minimised disturbance to the plot from the operator, vacuum air vent, and engine exhaust (Trumble *et al.* 1981). Upon completion of the plot, the end of the vacuum tube was immediately blocked with the plug in order to prevent arthropods from escaping and/or suctioning additional arthropods outside of the sampling period (Ozanne 2005). The collection bag was removed while the suction was still applied (Buffington and Redak 1998), closed with a wire tie, and placed in a large killing bottle charged with ethyl acetate until the arthropods were dead (Triplehorn and Johnson 2005). Samples were transferred to plastic bags and frozen for storage at -20° C until sorting, in order to preserve nutrient and energetic contents for biomass measurements (Robel *et al.* 1995). At each quadrat the vertical structure of vegetation was measured with a collapsible 0.8 x 200 cm steel pole marked at 1 cm increments and fitted with a 10 cm plastic drop-plate. Mean vegetation height (cm) was calculated by taking drop-plate measurements at five systematic points within each quadrat (Cherrill and Rushton 1993).



Fig. 1. Vacuum sampling eastern buttongrass moorland matrix habitat for arthropods at Bedlam (BED; 16 years post-fire), Lake St Clair, Tasmania (Photo: K. Cooper Chaudhry).

A variety of different methods are often used to separate invertebrates samples from plant and other material such as UV light extraction (Buffington and Redak 1998), sieving (Dennis *et al.* 1998), flotation (Ausden 1996), and phase-separation techniques (Barmuta 1984). Since vacuum sampling often results in large volumes of organic matter that can be very time-consuming and tedious to sort through by hand (i.e. $\sim \leq 500$ ml per quadrat for this study), a number of pilot studies were conducted to evaluate the appropriateness and effectiveness of live and dead extraction techniques for this study. A formal pilot was conducted following the methods of Buffington and Redak (1998) and the extraction efficiencies of four trials were determined by calculating the percentage of arthropods extracted into the collection head based on the total number of dead specimens identified in the sample by hand sorting. Extraction efficiencies ranged from only 13-26%, were biased towards positively phototactic and flying insects, and showed notable intraorder and interorder variability in extraction success. Informal trials were also conducted on the latter methods which were unsuccessful in separating the arthropods from the organic material. Accordingly, all samples were hand sorted by the author to maximise the recovery of arthropods from the samples and minimise intersorter bias (Barmuta 1984). Despite the potential unreliability of this method (Barmuta 1984), hand sorting is typically considered to be less biased overall than other sorting techniques (Ausden 1996; Southwood and Henderson 2000).

Frozen samples were allowed to thaw and were thoroughly examined with a dissecting microscope at 10x magnification in a sorting tray. All specimens were identified to order using the identification keys of Zborowski and Storey (1995), New (1996), Harvey and Yen (1997), and Naumann (1991), and nymphs and larvae were recorded separately. Counting was focused on bodies, and moults and specimens that had clearly died prior to sampling were not recorded, although in practice this can sometimes be difficult to determine (Southwood and Henderson 2000). Detached heads and appendages (i.e. of a reasonable size, e.g. Orthoptera legs) that could be identified were included in vials of their respective orders. Total and mean abundance (\pm SE) were calculated for each of the potential prey orders for each sample.

Calculating food availability solely based on relative abundance can be misleading (Beaver and Baldwin 1975; Johnson 2000; Buchanan *et al.* 2006). Therefore, individuals were pooled by order within each sample and weighed within the microvials as the large sample size prohibited weighing or measuring specimens individually. All vials were opened and placed in drying racks in an oven at 50° C for 72 hours, after which point there was no decrease in weights. Each vial was weighed to the nearest 0.1 mg using an analytical balance (Mettler AE 200, ± 0.1 mg SD). The original vials were emptied, cleaned, and

weighed to determine the individual tare weights for each vial. Dry weight biomass (mg DW m⁻²) was then converted to a more biologically meaningful estimate of gross energy content (Joules m⁻²) by using published conversion factors for each order (Table 8).

Arthropod prey analyses

As there was no *a priori* ecological interest in variation at the quadrat scale, mean values (\pm SE) of abundance (no. m⁻²), biomass (mg Dry Weight m⁻²) and energy content (J m⁻²) were calculated by order ($n = 242$) and pooled across orders ($n = 22$) from the three quadrats within each habitat and site. In the case of the former, although the mean number of arthropods was based on 1 x 1 m quadrats and hence could be considered an estimate of density, due to the different volumes of vegetation sampled within each quadrat, it is more appropriately referred to as a (relative) estimate of abundance. Absences in abundance, biomass, and energy within quadrats were treated as true zero values. Two-tailed Spearman rank correlations indicated that both mean abundance ($r_s = 0.714$; $n = 22$; $P = 0.004$) and mean biomass ($r_s = 0.995$; $n = 22$; $P < 0.001$) were correlated with mean energy content when pooled across orders within each habitat and site. Therefore, mean biomass was excluded from further univariate analysis; however, both mean abundance and mean energy content were included as they are often considered to be complementary measures of prey availability (e.g. Bryant 1973, Maguire 2006b). Similarly, two-tailed Spearman rank correlations indicated that both mean abundance ($r_s = 0.906$; $n = 242$; $P < 0.001$) and mean biomass ($r_s = 0.998$; $n = 242$; $P < 0.001$) were highly correlated with mean energy content by order within each habitat and site and were therefore excluded from further multivariate analyses. Total abundance and total energy content per order were used to estimate the mean energy content per individual (J individual⁻¹); although, as specimens were not weighed individually no standard error terms are presented and these figures are only provided for illustrative purposes.

A split-plot mixed model analysis of variance (ANOVA) was used to test for overall significant differences in mean arthropod prey abundance (no. m⁻²) and mean energy content (J m⁻²) of quadrats (pooled across orders) in response to habitat and fire age using SYSTAT 10 (SPSS Inc. 2000). Separate tests were conducted for the two dependent variables, with habitat type and fire age as fixed factors, and the site replicates (nested within fire age) treated as a random factor. Although the sites were chosen based on the selection criteria described in Chapter 2, they are included as a random factor as they are considered to be representative of their respective fire ages in eastern moorlands and as it is consistent with the original intent of the study design and hypotheses (G. Quinn pers. comm. 2006). The residual term (quadrats) was omitted from the models as the associated hypotheses were not of interest. Fire age three (HAR, see Chapter 2) was excluded from these analyses as only

one site was available for sampling (i.e. no replication). The dependent variables were not transformed as the assumptions of normality and homogeneity of variances were met. Furthermore, examination of Studentised residuals did not indicate any issues of concern regarding outliers and model fit. *Post hoc* tests using Tukey HSD were conducted to test for significant differences between fire ages. Two-tailed Spearman rank correlations were conducted to determine whether there were any significant correlations between mean vegetation height and the dependent variables.

The influence of arthropod community structure on observed patterns in energy content and distribution was explored using a range of multivariate techniques available in Primer 5.2.2 (Primer-E Ltd. 2001). The similarities of mean energy content contributed by orders between different samples were calculated using the Bray-Curtis similarity measure as it ignores joint absences and is often recommended for community data (Quinn and Keough 2002; McCune *et al.* 2002). The data were untransformed as there was no interest in reducing the influence of the orders with the greatest biomass (i.e. energy content) (Clarke and Gorley 2001). The similarity matrix was analysed with non-metric Multidimensional Scaling (MDS) using two and three dimensions with 30 random restarts; however, for ease of display and interpretation the best 2-dimensional configuration with its associated stress was chosen (Clarke and Warwick 1994). The habitat and fire age factors were plotted on the ordination plot to show any apparent patterns with the underlying similarities. A cluster analysis (CLUSTER) using a group mean and ranked similarities was also performed on the Bray-Curtis similarity matrix and five clusters (arbitrarily identified at the 120 rank level) were superimposed on the MDS plot to assess the adequacy and consistency of the MDS and CLUSTER methods (Clarke and Warwick 1994). The similarity percentages procedure (SIMPER) was conducted to examine the mean one way similarities within and mean one way dissimilarities between the habitat and fire age factors and thus determine which orders were contributing the most to the patterns observed in the MDS plot (Clarke and Gorley 2001). As complex designs such as split-plots cannot be accommodated in analysis of similarities (ANOSIM) procedures, one way ANOSIMs (999 random permutations) were used to test for significant differences in the energy contributions of different arthropod orders between habitats and between fire ages among the 11 sites as depicted in the MDS plot (Clarke and Gorley 2001). *Post hoc* tests using a Bonferroni adjusted *P*-value were conducted to test for significant differences between fire ages.

Results

Habitat use

Observations of habitat use in the study area ($n = 250$) were dominated by the resident Southern Emu-wren (17%) and Striated Fieldwren (34%), while the Ground Parrot was rarely observed (2%). The 18 non-resident species together accounted for the remaining 47% of observations. The results of the log-linear modelling in relation to groups are summarised in the analysis of deviance shown in Table 2 and their parameter estimates are outlined in Table 3. The saturated models provided the best fit for the data at Lake Pedder and Lake St Clair and demonstrated significantly non-random selection of habitats ($P < 0.001$) that depended on the availability of different habitats, group membership, and their interactions. The estimated relative selection probabilities (Table 4) indicate the relative expected use if habitats were available in equal proportions (Garshelis 2000). In all cases, the birds used riparian and edge habitats more than expected (range $\hat{w} = 1.57$ -42.00), with the large values of these probabilities resulting from the extensive use (48%, $n = 250$) and the very limited availabilities of these habitats ($\leq 5\%$ of total area each within either location). Overall, the probabilities of selection for the non-matrix habitats were relatively higher in the sedge-dominated eastern moorlands at Lake St Clair than the scrub-dominated blanket moorlands at Lake Pedder. This was particularly the case in relation to the apparent preference for riparian habitats demonstrated by both the resident and non-resident groups at Lake St Clair ($\hat{w} = 15.04$ and 21.07, respectively). As expected, the residents accounted for the majority (70%) of total observations in the moorland matrix ($n = 133$); however, 39% of resident observations in the matrix ($n = 92$) were originally noted as being in scrub copses, while 65% ($n = 40$) of non-resident observations were in scrub copses or isolated trees. Residents also appeared to weakly select edge habitat over matrix at both locations, although the effect appeared to be stronger at Lake Pedder ($\hat{w} = 4.60$ vs. 1.57). However, it should be noted that of the three resident species, only the Striated Fieldwren was recorded using edge habitat ($n = 6$), and in all of these cases the edge habitat was originally noted as being primarily comprised of scrub species (e.g. *Leptospermum* and *Melaleuca* spp.) associated with ecotones between the moorland and woodland or along road edges. In this sense, the apparent selection of edge habitats by the residents was again largely an artefact of the necessary scale of mapping and habitat classification (see Chapters 2-3), as opposed to a demonstrated affinity for the typically *Eucalyptus*-dominated edges *per se*. Furthermore, a lack of any off-survey observations within the woodlands helped to substantiate the contention that the resident species do not utilise woodland habitats within the study area. In contrast, the relatively large probabilities of non-resident species using edge habitat at Lake Pedder and Lake St Clair ($\hat{w} = 38.64$ and 42.00, respectively) helped to confirm their *a priori* categorisation. In other words, the non-resident group included species typically associated with adjacent woodland habitats that primarily fall outside of the moorland sites as defined.

The results of the log-linear modelling in relation to fire age are summarised in the analysis of deviance shown in Table 5 and the parameter estimates from the models of best fit are outlined in Table 6. The results demonstrated significantly non-random selection of habitats at both locations ($P < 0.001$). At Lake St Clair, the saturated model included a significant interaction between habitat and fire age ($P = 0.002$) and therefore cannot be reduced any further, indicating that availability of different habitats, fire age, and their interaction were significantly related to habitat use. In contrast, at Lake Pedder the reduced model provides the best fit ($P < 0.001$) indicating that fire age was not significantly related to habitat selection. As in the group analysis (Table 4), the relative selection probabilities (Table 7) indicated that the avifauna used riparian and edge habitats more than expected in all cases (range $\hat{w} = 5.02$ -79.48). When compared to all ages at Lake Pedder, the relative probabilities for the riparian and edge habitats were notably higher across fire ages at Lake St Clair, with the exception of the 3 year old sites. In this sense the models were somewhat redundant, particularly for Lake Pedder as the reduced model necessitated the pooling of relative selection probabilities across fire ages and essentially provided a community-level summary of those presented in Table 4. However, as fire age was significantly related to habitat selection at Lake St Clair some important inferences can be made. The most striking pattern was at the one year old site (HAR), where there was a strong apparent preference for riparian habitat ($\hat{w} = 79.48$), most of which was largely unburnt by the prescribed fire in October 2003 (pers. obs.; Figure 2). Although it was evident that the magnitude of this preference was considerably reduced in older sites (range $\hat{w} = 3.70$ -41.45), there was no clear linear trend across the chronosequence as indicated by these probabilities and the coefficient estimates.



Fig. 2. Partially unburnt riparian zone through burnt eastern moorland matrix (approx. drainage centerline as marked), one day post-fire, Harbacks Road (HAR), Lake St Clair, Tasmania.

Table 2. Analysis of deviance table for the log-linear models of habitat selection in relation to the resident and non-resident birds of buttongrass moorlands at Lake Pedder ($n = 108$) and Lake St Clair ($n = 142$), Tasmania. * Denotes significance at the 0.05 and ** at the 0.01 level and ^ denotes model of best fit.

Model by location	df	Deviance	df	Residual deviance	P
Lake Pedder					
No selection of habitat			5	96.025	
Selection by habitat	2	69.685	3	26.341	< 0.001**
Selection by habitat + group	1	4.513	2	21.828	0.034*
Selection by habitat + group + habitat x group ^	2	21.828	0	< 0.001	< 0.001**
Lake St Clair					
No selection of habitat			5	276.789	
Selection by habitat	2	241.301	3	35.488	< 0.001**
Selection by habitat + group	1	0.451	2	35.037	0.502
Selection by habitat + group + habitat x group ^	2	35.037	0	< 0.001	< 0.001**

Table 3. Parameter estimates from the log-linear models of best fit for habitat selection in relation to the matrix habitat and resident bird group of buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. * Denotes significance at the 0.05 and ** at the 0.01 level.

Coefficient by location	Estimate	SE	Z	P
Lake Pedder				
Constant	4.079	0.135	30.249	< 0.001**
Riparian	0.627	0.430	1.459	0.145
Edge	1.526	0.518	2.946	0.003**
Non-residents	-1.117	0.272	-4.113	< 0.001**
Riparian x non-residents	1.964	0.558	3.518	< 0.001**
Edge x non-residents	2.129	0.644	3.306	< 0.001**
Lake St Clair				
Constant	3.715	0.162	22.902	< 0.001**
Riparian	2.711	0.252	10.770	< 0.001**
Edge	0.450	0.726	0.621	0.535
Non-residents	-0.547	0.268	-2.040	0.041*
Riparian x non-residents	0.342	0.393	0.870	0.384
Edge x non-residents	3.287	0.777	4.230	< 0.001**

Table 4. Relative selection probabilities (\hat{w}) in relation to the resident and non-resident groups estimated from the resource selection functions for the avifauna of buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. Availability of matrix habitat is standardised as 1.00.

Habitat by location		Group	
	Lake Pedder	Residents	Non-residents
Matrix		1.00	1.00
Riparian		1.87	13.35
Edge		4.60	38.64
	Lake St Clair		
Matrix		1.00	1.00
Riparian		15.04	21.17
Edge		1.57	42.00

Table 5. Analysis of deviance table for the log-linear models of habitat selection in relation to fire age by the avifauna of buttongrass moorlands at Lake Pedder ($n = 108$) and Lake St Clair ($n = 142$), Tasmania. * Denotes significance at the 0.05 and ** at the 0.01 level and ^ denotes model of best fit.

Model by location	df	Deviance	df	Residual deviance	P
Lake Pedder					
No selection of habitat			11	89.332	
Selection by habitat ^	2	71.350	9	17.982	< 0.001**
Selection by habitat + fire age	3	7.018	6	10.964	0.071
Selection by habitat + fire age + habitat x fire age	6	10.964	0	< 0.001	0.089
Lake St Clair					
No selection of habitat			14	296.576	
Selection by habitat	2	243.271	12	53.305	< 0.001**
Selection by habitat + fire age	4	29.415	8	23.890	< 0.001**
Selection by habitat + fire age + habitat x fire age ^	8	23.890	0	< 0.001	0.002**

Table 6. Parameter estimates from the log-linear models of best fit for habitat selection in relation to the matrix habitat and fire age class 1 (1 year post-fire at Lake St Clair) by the avifauna of buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. * Denotes significance at the 0.05 and ** at the 0.01 level.

Coefficient by location	Estimate	SE	Z	P
Lake Pedder				
Constant	2.972	0.117	25.396	< 0.001**
Riparian	1.613	0.252	6.389	0.028*
Edge	2.558	0.284	9.022	< 0.001**
Lake St Clair				
Constant	1.437	0.500	2.873	0.004**
Riparian	4.376	0.584	7.494	< 0.001**
Edge	2.686	1.118	2.403	0.016*
Fire age 3	0.967	0.592	1.634	0.102
Fire age 5-8	1.818	0.540	3.366	< 0.001**
Fire age 14-16	0.588	0.627	0.938	0.348
Fire age 30-31	1.349	0.563	2.398	0.016*
Riparian x fire age 3	-3.068	0.970	-3.163	0.002**
Edge x fire age 3	-1.339	1.360	-0.984	0.325
Riparian x fire age 5-8	-1.912	0.674	-2.838	0.005**
Edge x fire age 5-8	-0.518	1.180	-0.044	0.965
Riparian x fire age 14-16	-0.651	0.745	-0.874	0.382
Edge x fire age 14-16	0.706	1.240	0.569	0.569
Riparian x fire age 30-31	-1.856	0.730	-2.543	0.011*
Edge x fire age 30-31	0.440	1.181	0.372	0.710

Table 7. Relative selection probabilities (\hat{w}) in relation to fire age estimated from the resource selection functions for the avifauna of buttongrass moorlands at Lake Pedder and Lake St Clair, Tasmania. Availability of matrix habitat is standardised as 1.00.

Habitat by location	Fire age (years post-fire)				
Lake Pedder		All (3-54)	—	—	—
Matrix		1.00			
Riparian		5.02			
Edge		12.90			
Lake St Clair		1	3	5-8	14-16
Matrix		1.00	1.00	1.00	1.00
Riparian		79.48	3.70	11.75	41.45
Edge		14.67	3.85	13.93	29.72
					22.78

Arthropod prey resources

A total of 15,861 arthropods were recorded with a dry mass of 3,498 mg and estimated energy content of 78,652 J, representing 9 potential prey orders (inclusive of larvae). Potential prey orders included in the analyses are as follows, with the percentage of the 1 m² quadrats in which they were present in parenthesis ($n = 66$): Araneae (100%), Blattodea (21%), Coleoptera (65%), Coleoptera larvae (20%), Diptera (100%), Hemiptera (98%), Hymenoptera (97%), Lepidoptera (41%), Lepidoptera larvae (55%), Orthoptera (27%) and Psocoptera (55%). Although included in the above figures, Diptera larvae and Neuroptera were not included in the analyses because they were only present in < 5% of samples (McCune *et al.* 2002), and were not deemed to be biologically significant. Hemiptera nymphs were also excluded from analyses as they were not identified as potential prey items in the literature and were unlikely to be the focus of foraging, at least in the case of lepp-forming psyllids (Woinarski *et al.* 1989). The only known arthropod prey orders that were not collected by vacuum sampling include Chilopoda, Dermaptera, and Mantodea; however, the latter has not been recorded in the TWWHA (Mallick and Driessen 2005). The following non-prey invertebrate taxa were also collected in the vacuum samples: Acarina (100%), Amphipoda (24%), Collembola (98%), Diplopoda (14%), Gastropoda (18%), Hemiptera nymphs (12%), Isopoda (18%), Mecoptera (17%), Neuroptera larvae (3%), Plecoptera (2%), Thysanoptera (76%), Trichoptera (20%), and Trichoptera larvae (2%). Only Acarina and Collembola were found in consistently high estimated relative abundance within quadrats, while the remaining orders were typically found in low relative abundance. Greenslade and Smith (1999) likewise reported that Acarina and Collembola were the most abundant orders in their inventory of epigaeic arthropods in Tasmanian buttongrass moorlands.

A summary of mean abundance, biomass, and energy content m⁻² and estimated mean energy content per individual for the prey orders is provided in Table 8. Mean figures for abundance, biomass, and energy content were notably higher in riparian than in matrix habitats for all orders except Orthoptera. Estimated energy content per individual ranged from 0.8 J individual⁻¹ for Psocoptera and Hymenoptera to 63.8 J individual⁻¹ for Orthoptera, reflecting a broad range of energy contents and associated sizes of available prey items (i.e. ~ < 1 mm to > 20 mm in length). In contrast to other trends, the estimated mean J individual⁻¹ was higher for the matrix than riparian habitats (6.3 vs. 4.3 J individual⁻¹).

Table 8. Measures of potential arthropod prey orders of insectivorous birds in buttongrass moorlands at Lake St Clair, Tasmania. Mean (\pm SE) abundance, biomass (mg Dry Weight), and energy content (J) of arthropod prey orders m⁻² vacuum sample ($n = 33$) and estimated mean energy content individual⁻¹ in matrix (MTX) and riparian (RIP) habitats pooled across buttongrass moorland sites ranging from < 1-32 years post-fire. Energy content conversions factors are from the literature (^aCummins and Wuycheck 1971, *excluding Stratiomyidae since rare in buttongrass moorlands (M. Driessen unpublished data); ^bO'Farrell *et al.* 1971 cited in Bell 1990; ^cBrodmann and Reyer 1999; ^dNorberg 1978). ^ Standard errors are not presented for J individual⁻¹ as these figures are derived estimates and not based on individual DWs of specimens.

Order	Habitat	<i>n</i>	Total abundance	Mean abundance	Mean biomass (mg DW)	Mean energy content (J)	Est. individual energy content ^ (J individual ⁻¹)	Energy conversions (J mg ⁻¹ DW)
Aranaeae	MTX	33	1254	38.0 \pm 4.2	10.7 \pm 1.7	216.2 \pm 34.8	5.7	20.19 ^a
	RIP	33	1613	48.9 \pm 7.0	11.3 \pm 1.4	227.2 \pm 28.6	4.6	
Blattodea	MTX	33	5	0.2 \pm 0.1	0.0 \pm 0.0	0.8 \pm 0.4	5.4	22.58 ^a
	RIP	33	14	0.4 \pm 0.1	0.5 \pm 0.3	11.5 \pm 5.8	27.1	
Coleoptera	MTX	33	39	1.2 \pm 0.3	1.8 \pm 1.1	41.8 \pm 25.8	35.4	23.25 ^a
	RIP	33	146	4.4 \pm 0.9	3.7 \pm 0.9	87.1 \pm 20.7	19.7	
Coleoptera larvae	MTX	33	1	0.0 \pm 0.0	0.0 \pm 0.0	0.2 \pm 0.2	5.5	27.56 ^b
	RIP	33	32	1.0 \pm 0.4	0.2 \pm 0.1	5.9 \pm 2.7	6.1	
Diptera	MTX	33	1617	49.0 \pm 13.0	8.1 \pm 2.0	185.4 \pm 44.9	3.8	22.78 ^{a*}
	RIP	33	5040	152.7 \pm 36.1	19.1 \pm 4.1	434.3 \pm 94.5	2.8	
Hemiptera	MTX	33	1538	46.6 \pm 6.0	18.3 \pm 2.2	432.1 \pm 52.7	9.3	23.59 ^a
	RIP	33	1805	54.7 \pm 9.7	22.8 \pm 3.9	537.3 \pm 92.5	9.8	
Hymenoptera	MTX	33	431	13.1 \pm 1.8	0.6 \pm 0.1	12.1 \pm 1.7	0.9	19.37 ^a
	RIP	33	1405	42.6 \pm 9.7	1.7 \pm 0.3	32.9 \pm 5.9	0.8	
Lepidoptera	MTX	33	16	0.5 \pm 0.1	0.5 \pm 0.4	10.9 \pm 8.7	22.4	22.42 ^c
	RIP	33	58	1.8 \pm 0.7	1.2 \pm 0.5	25.9 \pm 10.7	14.7	
Lepidoptera larvae	MTX	33	37	1.1 \pm 0.4	0.3 \pm 0.1	7.0 \pm 3.1	6.2	22.95 ^c
	RIP	33	74	2.2 \pm 0.5	1.4 \pm 0.8	32.5 \pm 19.0	14.5	
Orthoptera	MTX	33	33	1.0 \pm 0.4	2.1 \pm 0.9	47.7 \pm 20.0	47.7	22.18 ^a
	RIP	33	4	0.1 \pm 0.1	0.3 \pm 0.2	7.7 \pm 4.6	63.8	
Psocoptera	MTX	33	31	0.9 \pm 0.4	0.0 \pm 0.0	0.8 \pm 0.3	0.8	21.7 ^d
	RIP	33	662	20.1 \pm 7.3	1.2 \pm 0.3	25.4 \pm 5.6	1.3	
Total Means	MTX	363	5002	13.8 \pm 1.7	3.9 \pm 0.5	86.8 \pm 10.2	6.3	
	RIP	363	10853	29.9 \pm 4.3	5.8 \pm 0.7	129.8 \pm 15.6	4.3	

A summary of mean arthropod abundance m^{-2} pooled across orders is presented in Figures 3 and 4 and the associated results from the split-plot ANOVA are presented in Table 9.

Results indicated that habitat was a significant factor ($P = 0.004$) while fire age was not a significant factor ($P = 0.219$). There was no significant interaction between habitat and fire age ($P = 0.295$). There was also no significant added component of variance from the nested site factor ($P = 0.219$). Two-tailed Spearman rank correlations indicated that mean abundance was significantly correlated with mean vegetation height ($r_s = 0.563$; $n = 22$; $P = 0.006$).

A summary of mean arthropod energy content m^{-2} pooled across orders is presented in Figures 5 and 6 and the associated results from the split-plot ANOVA are presented in Table 10. Results for the main effects were in contrast to those described for abundance, with habitat identified as marginally non-significant ($P = 0.051$) and fire age as a significant factor ($P = 0.038$). *Post hoc* tests revealed that there was a significant difference in mean arthropod energy content between fire ages 1 and 16 ($P = 0.049$), with the next smallest P values between 1 and 3 ($P = 0.098$) and 16 and 32 ($P = 0.088$). While habitat was marginally non-significant, the relatively high F -ratio ($F = 6.498$) and low degrees of freedom ($df = 1$) indicated that habitat appeared to have some ecological effect on the response variable, with riparian habitats typically being higher in energy content than matrix habitats across all fire ages. Similar to the results above, there was no significant interaction between habitat and fire age ($P = 0.445$) and again the nested site factor was not significant ($P = 0.512$), indicating the sites were serving as reasonable replicates of their respective fire ages. Contrary to the above results, two-tailed Spearman rank correlations indicated that mean energy content was not significantly correlated with mean vegetation height ($r_s = 0.256$; $n = 22$; $P = 0.251$).

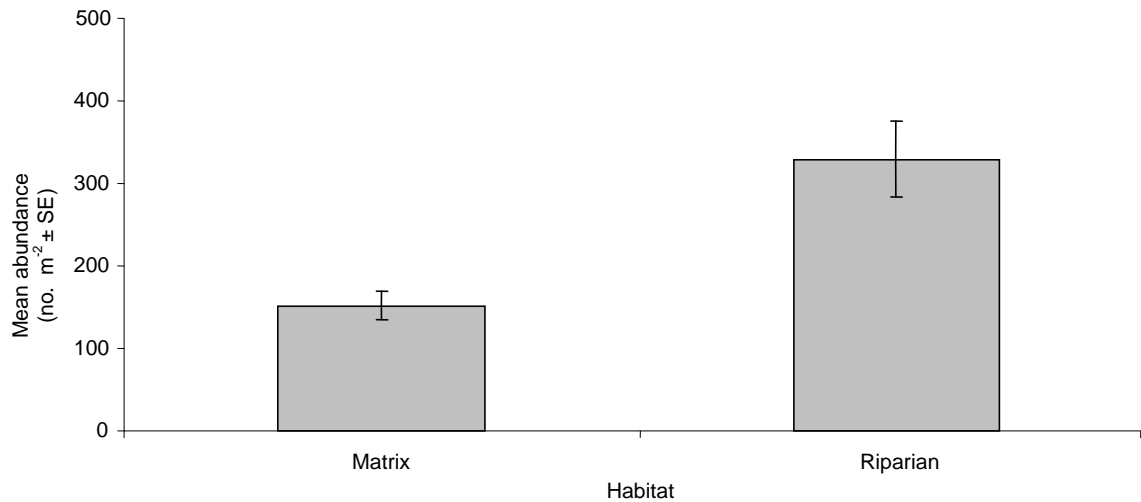


Fig. 3. Mean abundance (no. m⁻² ± SE) of potential arthropod prey (pooled across orders and fire ages) in matrix and riparian habitats of buttongrass moorlands at Lake St Clair, Tasmania ($n = 33$).

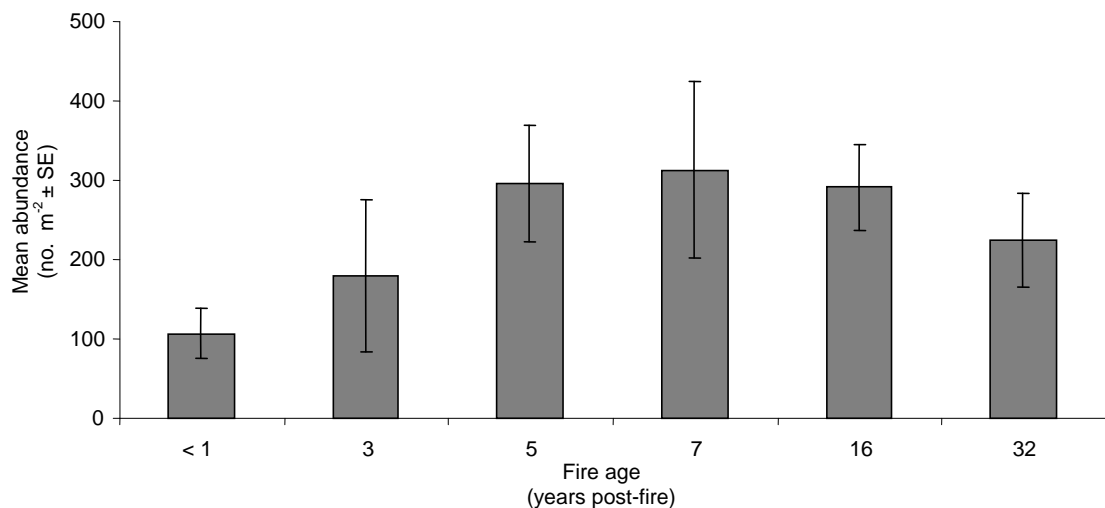


Fig. 4. Mean abundance (no. m⁻² ± SE) of potential arthropod prey (pooled across orders and habitats) in buttongrass moorlands ranging in age from < 1-32 years post-fire at Lake St Clair, Tasmania ($n = 12$ for all ages except age 3 where $n = 6$).

Table 9. Split-plot ANOVA for mean arthropod abundance (no. m⁻²) in response to habitat type (matrix and riparian) and fire age (< 1, 5, 7, 16, 32 years post-fire) in buttongrass moorlands at Lake St Clair, Tasmania. * Denotes significance at the 0.05 level.

Variable	SS	df	MS	<i>F</i>	<i>P</i>
Habitat	1.548 E+05	1	1.548 E+05	23.954	0.004*
Fire age	1.150 E+05	4	2.875 E+04	2.092	0.219
Habitat x fire age	4.272 E+04	4	1.068 E+04	1.652	0.295
Site (Fire age)	6.871 E+04	5	1.374 E+04	2.126	0.214
Error	3.232 E+04	5	6.463 E+03		

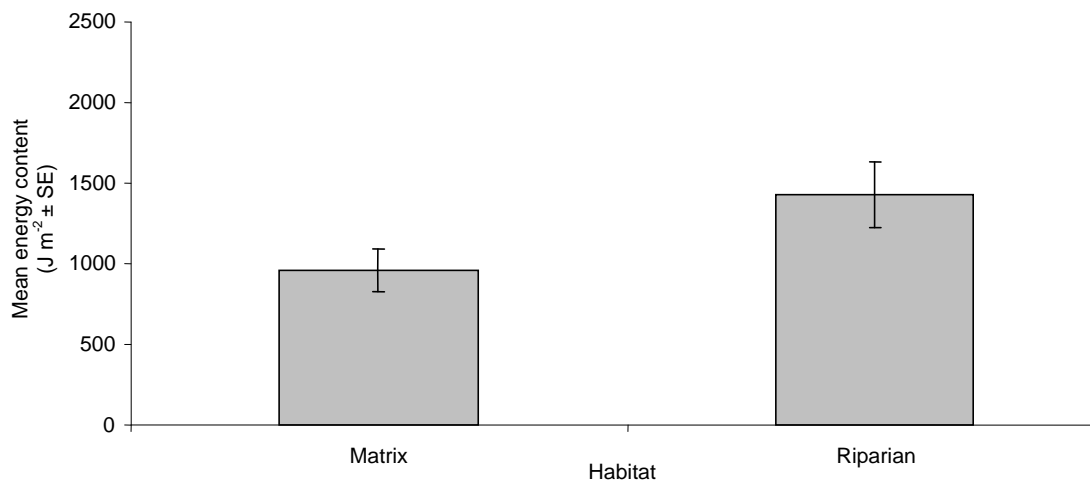


Fig. 5. Mean energy content ($\text{J m}^{-2} \pm \text{SE}$) of potential arthropod prey (pooled across orders and fire ages) in matrix and riparian habitats of buttongrass moorlands at Lake St Clair, Tasmania ($n = 33$).

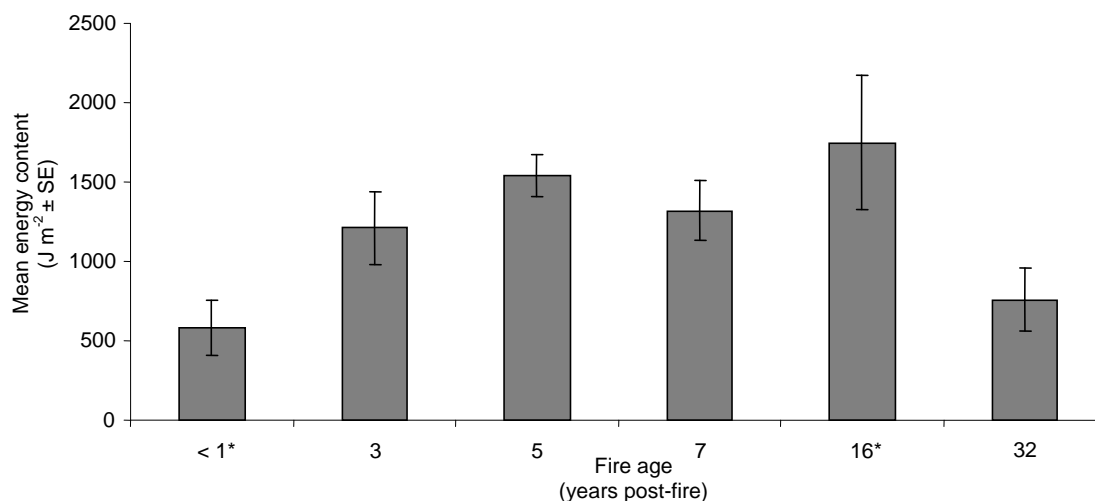


Fig. 6. Mean energy content ($\text{J m}^{-2} \pm \text{SE}$) of potential arthropod prey (pooled across orders and habitats) in buttongrass moorlands ranging in age from < 1-32 years post-fire at Lake St Clair, Tasmania ($n = 12$ for all ages except age 3 where $n = 6$). * Denotes significant results from *post hoc* Tukey HSD tests ($P < 0.05$, excluding fire age 3).

Table 10. Split-plot ANOVA for mean arthropod energy content (J m^{-2}) in response to habitat type (matrix and riparian) and fire age (< 1, 5, 7, 16, 32 years post-fire) in buttongrass moorlands at Lake St Clair, Tasmania. * Denotes significance at the 0.05 level.

Variable	SS	df	MS	F	P
Habitat	1.123 E+06	1	1.123 E+06	6.498	0.051
Fire age	4.058 E+06	4	1.014 E+06	6.031	0.038*
Habitat x fire age	7.665 E+05	4	1.916 E+06	1.108	0.445
Site (Fire age)	8.411 E+05	5	1.682 E+06	0.973	0.512
Error	8.643 E+05	5	1.729 E+05		

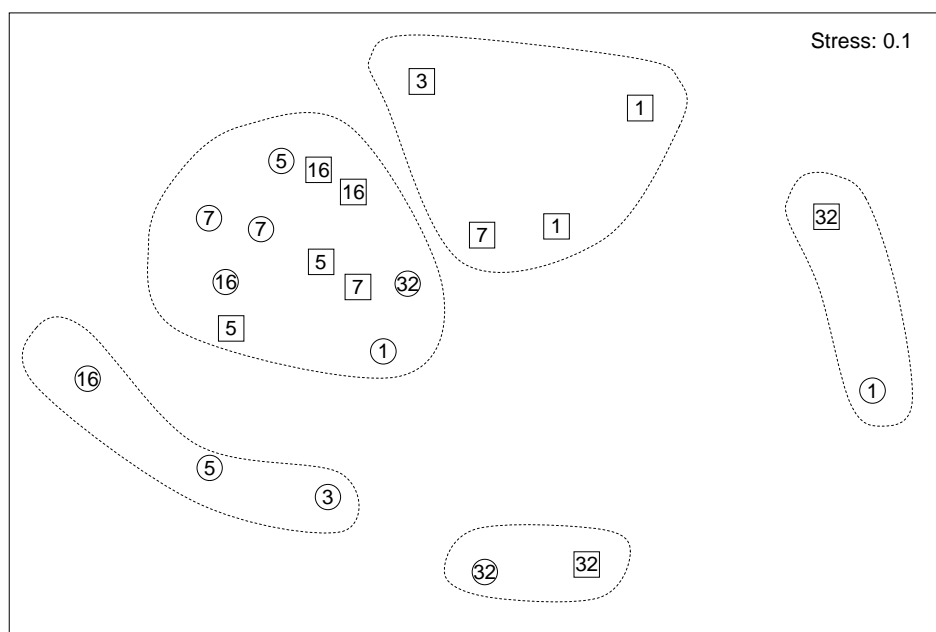


Fig. 7. Two-dimensional non-metric MDS ordination of the mean energy content (J m^{-2}) of arthropod prey orders in matrix (\square) and riparian (\circ) habitats at 11 replicate sites from < 1-32 years post-fire (#) at Lake St Clair, Tasmania. Group-mean clusters from Bray-Curtis ranked similarities (\cdots ; at an arbitrary level of 120) are superimposed to demonstrate the consistency between the two methods.

The non-metric MDS ordination plot is presented in Figure 7 along with the groups superimposed from the cluster analysis. The MDS had a stress value of 0.1, indicating a good two-dimensional ordination that can be interpreted with confidence (Clarke and Warwick 1994), and is at the low end of values that can be expected from most ecological community datasets (McCune *et al.* 2002). In addition, the groups identified from the cluster analysis were consistent with those that can be identified from the MDS plot at a range of rank levels, including that presented in Figure 7, providing additional confidence in subsequent interpretations (Clarke and Warwick 1994). Although there were no definitive and consistent clusters in relation to habitat and fire age, a few patterns emerged upon closer inspection. One group of three sites was composed of riparian habitats while another group of four sites was composed of matrix habitats, indicating there was moderate similarity within habitat types for some sites despite representing a range of fire ages. Furthermore, when habitats were compared in relation to fire age, similarities were exhibited between the same habitats from sites of the same age (e.g. within 16 year old matrix and within 7 year old riparian habitats), again indicating that these sites were serving as reasonable replicates of their respective ages. Conversely, there was also notable distance between different habitats from the same fire ages (e.g. < 1 and 3 year old matrix and riparian, respectively), indicating greater dissimilarity between habitats than between sites themselves. A notable exception to this pattern was the cluster of matrix and riparian habitats for the same 32 year old site (RCR), which was also noted as having the tallest mean height for the matrix (94.5 cm) and has a higher proportion of scrub species than any of the other sites sampled. The other group

composed of a < 1 year old riparian and 32 year old matrix sample also appeared to be somewhat of an outlier and represented the two sites with the lowest mean energy content. Finally, there appeared to be a grouping of both matrix and riparian habitats largely composed of middle aged sites ranging from 5-16 years, which was consistent with the pattern highlighted in Figure 6.

Results from the SIMPER analysis showed mean similarities of 56.24 for the matrix and 53.80 for riparian habitats, while their mean dissimilarity was 47.29 (Tables 11-12). Similarities of fire ages ranged from 35.36 for the 3 year old site to 71.12 for the 7 year old sites, while dissimilarities ranged from 30.64 between the 7 and 16 and year old sites and 57.52 between the 3 and 32 year old sites. These (dis)similarities were consistent with the general patterns presented in the MDS and cluster analyses, demonstrating some moderate although inconsistent clustering in relation to habitat and fire age. Hemiptera, Diptera, and Araneae dominated the community structure, together contributing over 80% of similarity within each habitat and fire age category and over 60% of mean dissimilarity in all pairwise comparisons within the habitat and fire age factors. Contributions from the remaining orders were relatively low and did not exceed 20% in any case. High ratios of the mean contribution of (dis)similarity of each order to their standard deviation indicate which orders typify samples as well as help to discriminate between samples in relation to these factors (Clarke and Warwick 1994). In most cases the ratios for these three orders were above the typical threshold of 1.4 for similarities within and to a lesser degree for dissimilarities between the categories, further indicating the important role they play in overall arthropod community patterns. This dominance was consistent with the fact that together they provided 85% of total energy sampled in the study and were ubiquitous in the study area, being present in both habitats at all sites. However, this was more a result of high abundances rather than large specimens, as on average (across habitats) the estimated energy content per individual for Hemiptera, Diptera, and Araneae (9.6, 3.1 and 5.1 J individual⁻¹, respectively) ranked at or below the median value (9.6 J individual⁻¹) (Table 8).

Table 11. Prey order contributions to the difference in energy content (J) between matrix and riparian habitats (mean similarity = 56.24 and 53.80, respectively) of buttongrass moorlands at Lake St Clair, Tasmania from SIMPER procedure (cut-off = 90%; see Table 8 for mean J by order and habitat).

Pairwise comparison (habitat)	Order	Mean dissimilarity	Dissimilarity/ SD	Contribution (%)	Cumulative contribution (%)
Matrix vs. Riparian (mean diss. = 47.29)	Hemiptera	16.20	1.36	34.25	34.25
	Diptera	13.32	1.30	28.17	62.42
	Araneae	6.15	1.21	13.01	75.43
	Coleoptera	3.82	0.88	8.09	83.52
	Orthoptera	2.24	0.64	4.74	88.26
	Lepidoptera larv.	1.47	0.49	3.11	91.37

Table 12. Prey order contributions to the difference in energy content (J) between fire ages of buttongrass moorlands at Lake St Clair, Tasmania from SIMPER procedure (cut-off = 80%). Fire ages (years post-fire) followed by mean similarity in parenthesis: < 1 (50.48), 3 (35.36), 5 (66.41), 7 (71.12), 16 (66.58), 32 (46.94).

Pairwise comparison (years post-fire)	Order	Mean dissimilarity	Dissimilarity/ SD	Contribution (%)	Cumulative contribution (%)
< 1 vs. 3 (mean diss. = 50.28)	Diptera	15.06	1.18	29.96	29.96
	Hemiptera	13.11	1.05	26.08	56.04
	Orthoptera	8.13	1.01	16.18	72.22
	Lepidoptera larv.	6.39	0.95	12.70	84.92
< 1 vs. 5 (mean diss. = 55.31)	Hemiptera	19.97	1.95	36.10	36.10
	Diptera	16.77	1.40	30.33	66.43
	Araneae	8.85	1.56	15.99	82.42
< 1 vs. 7 (mean diss. = 47.41)	Hemiptera	22.79	1.59	48.08	48.08
	Diptera	9.76	1.97	20.59	68.66
	Araneae	5.97	1.28	12.60	81.26
< 1 vs. 16 (mean diss. = 55.51)	Hemiptera	27.17	2.17	48.95	48.95
	Diptera	15.33	1.12	27.62	76.57
	Araneae	6.26	0.93	11.28	87.85
< 1 vs. 32 (mean diss. = 53.09)	Diptera	13.76	1.46	25.93	25.93
	Hemiptera	13.47	1.71	25.37	51.29
	Coleoptera	9.13	1.01	17.20	68.49
	Araneae	8.57	1.39	16.14	84.63
3 vs. 5 (mean diss. = 42.78)	Diptera	11.90	1.29	27.83	27.83
	Hemiptera	9.25	1.68	21.63	49.46
	Araneae	6.24	1.58	14.58	64.03
	Orthoptera	4.99	0.97	11.67	75.70
	Lepidoptera larv.	4.23	1.03	9.88	85.58
3 vs. 7 (mean diss. = 41.53)	Hemiptera	12.70	1.62	30.59	30.59
	Diptera	10.56	2.28	25.43	56.01
	Orthoptera	5.41	1.23	13.03	69.04
	Lepidoptera larv.	4.60	1.00	11.07	80.11
3 vs. 16 (mean diss. = 46.30)	Hemiptera	14.56	2.28	31.45	31.45
	Diptera	14.09	1.40	30.44	61.89
	Orthoptera	4.84	0.95	10.46	72.35
	Araneae	4.25	0.97	9.19	81.54
3 vs. 32 (mean diss. = 57.52)	Hemiptera	15.02	1.28	26.12	26.12
	Diptera	13.52	1.57	23.51	49.63
	Orthoptera	7.44	0.98	12.93	62.56
	Araneae	5.93	1.39	10.31	72.86
	Coleoptera	5.91	1.03	10.27	83.13
5 vs. 7 (mean diss. = 31.84)	Diptera	9.50	1.28	29.83	29.83
	Hemiptera	8.92	2.02	28.00	57.83
	Araneae	5.37	1.83	16.87	74.70
	Coleoptera	2.70	1.11	8.46	83.17

Table 12. cont.

Pairwise comparison (years post-fire)	Order	Mean dissimilarity	Dissimilarity/ SD	Contribution (%)	Cumulative contribution (%)
5 vs. 16 (mean diss. = 31.08)	Diptera	12.97	1.47	41.74	41.74
	Hemiptera	6.16	1.60	19.84	61.57
	Araneae	5.17	1.37	16.65	78.22
	Coleoptera	3.34	1.35	10.76	88.99
5 vs. 32 (mean diss. = 54.88)	Hemiptera	21.60	2.20	39.37	39.37
	Diptera	12.92	1.32	23.55	62.92
	Araneae	9.09	1.37	16.56	79.48
	Coleoptera	5.04	1.14	9.19	88.67
7 vs. 16 (mean diss. = 30.64)	Diptera	10.41	1.02	33.99	33.99
	Hemiptera	9.48	1.24	30.96	64.95
	Araneae	3.83	1.30	12.49	77.43
	Coleoptera	2.22	0.79	7.25	84.69
7 vs. 32 (mean diss. = 51.66)	Hemiptera	24.86	2.01	48.12	48.12
	Araneae	7.60	1.39	14.72	62.84
	Diptera	5.93	1.38	11.48	74.32
	Coleoptera	5.38	1.08	10.41	84.73
16 vs. 32 (mean diss. = 57.81)	Hemiptera	28.19	2.35	48.76	48.76
	Diptera	12.90	1.05	22.31	71.07
	Araneae	7.50	1.11	12.97	84.04

The global ANOSIM test showed there were borderline significant differences in arthropod community energy contributions between matrix and riparian habitats ($R = 0.089$, $P = 0.05$); however, the low R value indicated that these did not form distinct groups (Clarke and Gorley 2001). The global ANOSIM test showed there were significant differences between fire ages ($R = 0.22$, $P = 0.007$). Using a Bonferroni adjusted value of $P = 0.003$ for the pairwise tests there were no significant differences between fire ages, with the smallest P values between fire ages < 1 and 5 ($R = 0.448$, $P = 0.029$), < 1 and 16 ($R = 0.479$, $P = 0.029$), and 16 and 32 ($R = 0.521$, $P = 0.029$). However, the moderate R values indicated there were clear, although overlapping differences between these fire ages (Clarke and Gorley 2001). Overall, these results were consistent with the general patterns highlighted in the above analyses.

Discussion

Habitat use

The avifauna of buttongrass moorlands and adjacent habitats in Tasmania demonstrated significantly non-random habitat use within the study area. Both the resident and non-resident groups utilised riparian and edge habitats more than expected when compared to the moorland matrix. These patterns were related, either directly or indirectly, to differences in habitat type and availability, group membership, and their interactions. While the residents comprised the majority of observations in the matrix, both the residents and non-residents appeared to demonstrate fine-scaled selection of scrub copses and other emergent vegetation within the matrix; however, this could not be analysed in relation to availability due to the relatively coarse scale of mapping available (see Chapter 3). Such areas shared similar structure and floristics to the riparian and edge habitats, suggesting that the reported relative selection probabilities for non-matrix habitats may be underestimated in relation to those of the matrix. Although the relative selection probabilities indicated that the residents show some selection for edge habitats, especially at Lake Pedder, in all cases the Striated Fieldwren used areas that were intermediary in structure and floristics between the matrix and non-matrix habitats (i.e. ecotonal), and not the woodland edges *per se*. In addition, no off-survey observations were made of any of the residents utilising habitat beyond the moorland boundaries (i.e. woodland and forest). In a habitat mosaic in southeastern Australia that is comparable to this study, Baker *et al.* (2002) investigated patterns of bird density and species richness across heath-woodland edges. They classified the Southern Emu-wren as a heath specialist and ecotone-shy at two locations and ecotone-neutral at one location, although the Southern Emu-wren demonstrated limited use of the *Eucalyptus* woodland and edge habitat. They did not have sufficient data to include the Ground Parrot and Striated Fieldwren in their analyses; however, all records were limited to the heath. These findings are in general agreement with the conclusions of this study in that all three species are considered to be heath specialists. Nevertheless, some previous studies have reported the Striated Fieldwren using dry regrowth (0-12 years post-logging) and wet sclerophyll forests within some regions of Tasmania (Thomas 1979; Taylor *et al.* 1997) and the Southern Emu-wren using low closed *Eucalyptus* woodlands in southeastern Australia (Maguire 2006a) and regrowth Karri forests (2-3 years post-logging) in southwestern Australia (Wardell-Johnson and Williams 2000). These disparities may be due to real variation in relation to different subspecies and the relative availability and suitability of habitats, or may likewise be partially a result of the scale and detail of vegetation classification. Overall, it appears that within the study area the residents' home ranges were limited to the moorland itself and that within the moorland they used ecotonal areas and showed strong preference for riparian habitats within and bordering the matrix. This is the

first time that moorland riparian zones have been identified as being important habitats for the resident birds.

The non-residents demonstrated even stronger preference for the non-matrix habitats as compared to the resident species. Their use of edge habitat is consistent with the known habitat relationships for the non-residents. Despite the fact that it was expected they would show limited use of the matrix and intra-matrix riparian habitats, the results highlight that they demonstrated a strong selection for the riparian habitats. The association of some of the non-resident species with riparian habitats within moorlands has only been suggested by Brown *et al.* (1993) based on qualitative observations of the Dusky Robin, Beautiful Firetail, and Yellow-throated Honeyeater using taller creekside vegetation. However, neither the resident species nor numerous other non-resident species recorded in this study were identified in this group, particularly the Superb Fairy-wren and Crescent Honeyeater that both comprised large percentages of riparian observations. Baker *et al.*'s (2002) study included many of the same species classified as non-residents for this study, including the Crescent Honeyeater, which they classified as a wood specialist and ecotone-neutral, while the Superb Fairy-wren and New Holland Honeyeater were both classified as habitat generalists and ecotone-conspicuous. They found that species richness and density in the woodland were twice those of the heath, which they attributed to greater habitat complexity, and determined that none of the ecotone-conspicuous species were truly ecotonal. In this sense, their findings are in general agreement with this study in that the non-resident species may use heath and edge habitat, but are not solely reliant on the heath to meet all of their resource requirements. Thus, unlike the resident species, inferences regarding apparent habitat preferences by non-residents should be made with more caution as an unknown and presumably large proportion of their home ranges lie outside of the moorland sites as defined. Within the context of the moorlands and associated habitats, the non-residents appeared to view the matrix primarily as non-habitat. They only seemed to show finer-scaled selection for areas that contain suitable structure and floristics within the matrix (e.g. riparian zones and ecotonal areas), especially at Lake St Clair. Considering the almost total lack of prior research on the moorland avifauna, these findings helped to demonstrate the relative contributions of the different classes to the overall observed patterns of avifaunal habitat use versus availability. These results also demonstrated that the riparian zones within and bounding the moorland matrix were viewed as suitable habitat for many species that are more typically associated with woodland habitats.

At the community level, habitat selection was exhibited across moorlands at different stages of post-fire succession. Time-since-fire and its interaction with habitat were significant at Lake St Clair but not at Lake Pedder, which may be partly attributed to the latter

chronosequence having limited recently burnt sites. When compared to the composite relative selection probabilities at Lake Pedder, selection for the riparian and edge habitats was notably higher across fire ages at Lake St Clair. This was particularly the case in relation to the apparently strong preferences for riparian habitats, especially for the one year old site where the majority of observations were in the largely unburnt riparian zones that provided important post-fire refugia for the resident species. Although it was evident that the magnitude of this preference was considerably reduced in older sites, there was no clear linear trend regarding the decreasing importance of riparian zones with increasing time-since-fire. The extent to which these habitats were left unburnt during the most recent fire was unknown for most of the sites and may have partially influenced the relative suitability of habitats within each fire age (see Chapter 2). In general, the riparian habitats at Lake St Clair seemed to provide greater structural and floristic diversity across fire ages relative to the sedge-dominated matrix of eastern moorlands, whereas such differences were less pronounced between habitats in the scrub-dominated blanket moorlands of Lake Pedder. This is in spite of the fact that post-fire recovery of vegetation density takes significantly longer in the low productivity blanket moorlands compared to the moderate productivity eastern moorlands (i.e. > 10 vs. 5-6 years post-fire) (Driessen 1999; M. Driessen unpublished data; also see Chapter 3). Since Lake St Clair and Lake Pedder differ in respect to vegetation communities, soil fertility, and altitude, these factors were confounded within and between locations and the extent to which they influenced the observed patterns, especially in relation to fire, is unknown. In addition, since the bird group and fire age factors could not be analysed together due to sample size limitations, it was not possible to discriminate between their relative contributions in overall patterns of habitat selection.

Although no previous research has focused on avifauna habitat use versus availability in relation to fire in Tasmanian moorlands or related habitats on the mainland, a number of studies can provide some insight into the apparent preferences for riparian areas, particularly in recently burnt sites. Gellie (1980) stated that the Striated Fieldwren is capable of finding sufficient food and cover almost immediately after fire, and has been observed foraging on recently burnt moorlands in Tasmania, while the Southern Emu-wrens may remain if adequate patches of unburnt dense cover are available, although he did not provide any substantiating data. Others also contend that small unburnt or partly burnt patches of vegetation, such as along creeks, may be necessary for the Southern Emu-wren to persist in the post-fire environment (Recher and Christensen 1981; Pickett 2005). Gellie (1980) stated that if adequate unburnt vegetation is not available, then Southern Emu-wren breeding in Tasmanian moorlands could be precluded for five to seven years; however, he again provided no substantiating data. The Ground Parrot has occurred at sites approximately one year post-fire, when the estimated minimal projective foliage cover of 30% was attained at

most sites (Bryant 1994). However, Ground Parrots have been noted using sites soon after fires (Gellie 1980). Younger sites, especially with small unburnt copses of older and denser vegetation, may provide adequate cover and become temporarily crowded within larger burn areas (Meredith *et al.* 1984; Bryant 1991). The lower flammability of riparian vegetation, as derived from moorland fire behaviour and fuel models (Marsden-Smedley *et al.* 1999), may limit the extent and severity of fires in riparian areas except under extreme conditions, and thus serve as important unburnt refugia to local populations directly affected by the fire. Even burnt riparian areas can still provide greater relative cover and foraging resources than the matrix since the woody infrastructure is typically not fully consumed by the fire (pers. obs.). The importance of these areas may be proportional to the size of the fire, as the distance to unburnt moorlands may exceed the limited dispersal capabilities of species such as the Southern Emu-wren (Pickett 2000; Maguire and Mulder 2004).

Most small birds use structural characteristics to distinguish between habitats (Cody 1985), and the results from this study and the limited research available on the residents support this notion, although the underlying mechanisms are still not well understood. Vegetation structure may be the primary proximate factor that elicits habitat use and is related to ultimate factors that affect survival and reproduction (Block and Brennan 1993). Although previous research on the resident species is limited, most studies seem to be in agreement that habitat structure, particularly regarding vertical complexity, is the primary determinant of habitat suitability and use (Bryant 1991; Gosper and Baker 1997; Wilson and Paton 2004; Maguire 2006a). Maguire (2006a) found that at multiple fine scales the Southern Emu-wren demonstrated selection for habitats with dense vertical foliage density provided by graminoids and shrubs, and dense horizontal cover provided by medium to tall shrubs (~ 1 m). Wilson and Paton (2004) likewise found that habitats used by three South Australian subspecies were typically composed of low, dense shrub layers. Similarly, Striated Fieldwrens have been noted to occur in shrubbier sites with emergent shrubs (≤ 1 m) and may also utilise taller scrub edges (≤ 3 m) bordering such sites (Recher 1981; Gosper and Baker 1997), while Ground Parrots in Tasmania also require low, dense vegetation (Bryant 1991).

The structural characteristics of moorland riparian zones probably provide the avifauna with critical resources such as adequate cover, perches, and nesting sites, particularly for the cryptic resident species. Both Southern Emu-wrens and Striated Fieldwrens are known to seek denser cover when disturbed (Sharland 1981; Schodde 1982) and perch on top of emergent shrubs to investigate the disturbance before retreating again (Chandler 1912; Sharland 1981; Gosper and Baker 1997), behaviours that are confirmed by personal observations in the study area. Thus the apparent preference for riparian zones by the

residents may in part reflect predator avoidance (Maguire 2006a, 2006b) and evasive action taken by them in response to disturbance by the observer, as the shrub-dominated drainages provide better cover relative to the matrix. These habitats may also provide protection from adverse weather conditions, which are typical of the study area (BOM 2004; see Chapter 2). Similar to many passerines that inhabit open terrain and do not have aerial flight songs (Cody 1985), both species also frequently issue their territorial songs from prominent perches (Gosper and Baker 1997; Pickett 2000). Emergent shrubs with relatively robust growth forms such as *Leptospermum*, *Melaleuca*, and *Banksia* species characterise the moorland riparian zones. However, such behaviours that require emergent vegetation may not only influence their selection, but may likewise increase the birds' detectability and introduce an unknown source of bias. Lastly, riparian zones may provide the specific structural elements required for nesting. Maguire (2006a) reported that Southern Emu-wrens nested in plants that had dense crowns and provided adequate cover and structural support for the nest. However, these characteristics were not linked with the success of nests (Maguire and Mulder 2004). Although data on the Striated Fieldwren are much more limited, and the Ground Parrot only constituted a small percentage of resident observations, available evidence suggests that riparian zones may likewise provide the dense foliage and shrubs known to be used for nesting (Legge 1908; Dove 1916; Sharland 1981; Bryant 1991).

Arthropod prey resources

Mean abundance of potential arthropod prey resources sampled in buttongrass moorlands was significantly higher in riparian than in matrix habitats across fire ages, while mean energy content was marginally non-significant in relation to habitat. It was expected that variances would be high considering the fine scale of sampling (1 m²), limited replication, and that many arthropods have heterogeneous distributions at such scales (Cooper and Whitemore 1990). In this sense, and consistent with the significant result for abundance which is correlated with energy content, this more than likely represents an ecologically significant effect. Vertical vegetation structure was significantly correlated with mean abundance, which may simply reflect the larger volume of vegetation that was sampled in the quadrats with taller vegetation (e.g. in riparian zones; see Methods) or may be directly or indirectly related to other factors that may influence arthropod abundance such as food resources, resting sites, and microclimate. The significantly higher abundance of arthropods in the more structurally complex riparian habitats compared to the matrix is consistent with results of Dennis *et al.* (1998) who found that the diversity and abundance of arthropods in tussock grasslands in Scotland were likewise higher in more structurally complex vegetation. These findings lend support to the habitat heterogeneity hypothesis that predicts a positive relationship between these measures and structural heterogeneity (Hart and Horwitz 1991). However, as there was no significant correlation between mean height and mean arthropod

energy content, the underlying processes are difficult to elucidate and may be influenced by site-specific factors that are not accurately reflected in the habitat categories. Although woodland edge arthropod communities were not examined in this study, Driscoll (2005) reported that Coleoptera communities formed significantly distinct groups between forest (i.e. eucalypt and rainforest) and buttongrass habitats in the Lake Pedder area. Both abundance and species richness were significantly lower in buttongrass, which he primarily attributed to the water-logged peat soils, but stated they may also result from differences in nutrient levels and vegetation. These results, as well as predictions from the habitat heterogeneity hypothesis, suggest that other arthropod orders whose life cycles are similarly influenced by factors such as soil moisture may likewise be found in greater relative abundance and species richness along woodland edges than in the adjacent moorland matrix.

The availability of prey resources as indicated by both mean abundance and mean energy content is higher in riparian habitats and in medium aged sites (5-16 years post-fire), while prey resources appear to be severely limited in recently burnt sites (< 1 year post-fire) and may also be less available in older sites (32 years post-fire). However, only mean energy content varied significantly between fire ages, with significant *post hoc* comparisons between the 1 and 16 year old sites. In comparison, preliminary results from a collaborative SFT research project on terrestrial invertebrates conducted by the Biodiversity Conservation Branch (M. Driessen unpublished data) during summer revealed that there was a non-significant trend of increasing total mean numbers in sweep net samples collected from buttongrass moorland matrix habitat at sites 3-27 years post-fire at Lake Pedder and 1-20 years post-fire at Lake St Clair. There was no indication of a decline in mean numbers in older sites (i.e. 31-54 and 30-31 years post-fire, respectively). Results from pitfall samples collected from the matrix showed that mean numbers of invertebrates at sites 3 years post-fire were significantly higher than at sites 22-54 years post-fire at Lake Pedder, while at Lake St Clair there were no significant differences in mean numbers between fire ages. However, at the latter there appeared to be a marked increase in abundance 2-3 years post-fire compared to the 1 year old sites, which may be largely attributed to increased catches of orthopterans. However, as these are preliminary results and the analysis was conducted at a coarse taxonomic level, it is difficult to draw any definitive conclusions in relation to the aims of this study. In addition, since pitfall samples, and to a lesser degree sweep net samples, may be biased towards capturing active insects, these results may be a reflection of changes in post-fire invertebrate activity and not changes in mean counts or abundance *per se* (M. Driessen pers. comm. 2007). In a similar study conducted by Greenslade and Driessen (1999), they found that total numbers and morphospecies richness of epigaeic arthropods from sweep samples collected during summer were significantly different between fire ages in the matrix at Lake Pedder and Lake St Clair and that most taxa appeared

to be influenced by fire. Overall their data indicated that abundance was lowest in the regrowth sites (< 5 years post-fire), peaked in the medium age sites (11-19 years), and appeared to decline in the older sites (> 25 years), although the trend was more pronounced at Lake St Clair. Even though the successional time scales may differ, the findings from these studies are in general agreement with other invertebrate fire research around the world that follows a reduction and recovery response pattern concomitant with that of the vegetation community (Brown 1991; Whelan *et al.* 2002). This pattern is characterised by populations that decline during or shortly after fire, may take years to recover to peak abundances, and then may ultimately decline again as the vegetation community succeeds and becomes unsuitable for portions of the community (Greenslade 1997; Whelan *et al.* 2002).

These general patterns were reflected at the ordinal level as well, with the community composition expressed as mean energy content significantly different between fire ages and borderline significant between matrix and riparian habitats. There was moderate although inconsistent clustering of samples in relation to habitat, and the habitats within sites generally had high levels of dissimilarity due to the relatively higher energy content in riparian zones. In relation to fire, the middle aged sites (5-16 years post-fire) generally had high levels of similarity due to their higher energy content. These patterns were largely driven by Hemiptera, Diptera, and Araneae, which together comprised the majority of total energy content (85%). Both Greenslade (1997) and Greenslade and Smith (1999) likewise found that of the prey taxa identified in this study, Hemiptera, Diptera, and Araneae dominated total abundance from sweep samples and yellow pan traps in buttongrass moorlands (i.e. matrix) at Lake St Clair and Lake Pedder. Of the prey taxa, Greenslade and Driessen (1999) reported that Hemiptera, as well as Coleoptera, Neuroptera, Psocoptera, and Lepidoptera differed significantly in relation to fire age.

Successional changes in habitat structure may influence fauna by directly regulating the abundance and distribution of resources, or indirectly through changes in microclimate (Brown 1991). Vegetation density in moderate productivity moorlands can recover to pre-fire levels 4 to 5 years post-fire, which is consistent with the reported recovery of arthropods from the chronosequence at Lake St Clair. However, vegetation density may take 10 to 15 years to recover in the low productivity moorlands at Lake Pedder (Driessen 1999; M. Driessen unpublished data), and the post-fire recovery of arthropods (e.g. Collembola) may be correspondingly slower when compared to moderate productivity sites (Driessen and Greenslade 2004). While Greenslade and Driessen (1999) found similar reduction and recovery patterns of arthropods at both Lake Pedder and Lake St Clair, they noted that the pattern was most marked at Lake St Clair and attributed this disparity to differences in

vegetation communities, climate, and altitude; although soil fertility is a likely factor as well. However, as these studies used different methods and were limited to the matrix it is difficult to extrapolate their findings to prey resources at Lake Pedder. Although one might expect there to be a significant interaction between fire age and habitat, since there would tend to be a greater difference in structure between habitats in recently burnt sites due to the lower flammability attributes of riparian zones and edges (Marsden-Smedley *et al.* 1999), this does not appear to be the case for this study. Since moorland fires can often be patchy and as fine-scaled fire history maps were not available, the actual fire histories of a given plot may differ from the ages reported for the site as a whole, and thus possibly confound the results.

The relationship between avifaunal habitat use and food availability

The avifauna of buttongrass moorlands demonstrated strong selection for riparian and edge habitats in comparison to the moorland matrix, particularly in the eastern moorlands at Lake St Clair. Both structure and floristics are known to influence habitat selection through their effects on arthropod prey resources (e.g. availability and foraging substrates) (Wiens and Rotenberry 1981; Fleishman *et al.* 2003), and it is assumed that there is a strong correlation between sampled arthropods and the actual availability of bird food (Hutto 1980). Mean abundance and mean energy content of potential arthropod prey orders were significantly (statistically and/or ecologically) higher in riparian habitats and in medium aged sites. While abundance may represent the number of opportunities for predation within a habitat, energy content (or dry biomass) is probably a better overall measure of food resources as it is linked with profitability and ultimately affects survival and reproductive success (Beaver and Baldwin 1975; Brodmann *et al.* 1997, 1999; Johnson 2000). Species of Hemiptera, Diptera, and Araneae were most abundant and provided the majority of energy resources across habitats and fire ages. These orders are probably the primary sources of avian food in Tasmanian moorlands, with the possible exception of prey orders that may have been under-sampled by the vacuum (e.g. Orthoptera and Lepidoptera). The demonstrated selection for, and the greater availability of, food resources in riparian areas likely reflects a functional correspondence between these factors, with habitat type serving as either a direct resource or an indirect cue to elicit a settling response and foraging behaviour (Cody 1981; Hutto 1985; Wiens 1985). Previous studies have established that shrub-dominated habitats structurally and floristically comparable to moorland riparian zones (see Chapter 3) provide dense cover and a major foraging substrate for birds such as the resident Southern Emu-wren in Victorian heaths and shrublands (Maguire 2006a). Riparian habitats support a greater abundance and energy content of arthropod orders that have been shown to be preferred by adult Southern Emu-wrens for the provisioning of nestlings, including Lepidoptera, Diptera, Hemiptera, and larvae (Maguire 2006b). Riparian habitats also provide a greater abundance of nectar-producing shrubs (e.g. *Epacris* spp., *Callistemon* spp., *Banksia* spp.) that are relied upon by

some of the nectivorous non-residents, such as the New Holland and Crescent Honeyeaters (Paton 1982; Higgins *et al.* 2001). Although edge habitats could not be included in this study, their closer structural and floristic affinities with riparian habitats suggest that they may likewise provide greater arthropod prey resources (Driscoll 2005) and more suitable foraging substrates, particularly for the non-resident species that appear to prefer, and are better adapted to, foraging in habitats dominated by scrub and tree species (Moermond 1990; Higgins *et al.* 2001; Higgins and Peter 2002). Finally, considering that arthropod sampling could not be conducted at Lake Pedder it is difficult to make any definitive statements; however, the habitat use results herein as well as results on arthropod sampling from previous studies suggest that similar patterns and processes may be taking place in blanket moorlands.

Habitat type, fire age, and their interaction had a significant influence on habitat use by the avifauna at Lake St Clair. Although there are no clear trends across the chronosequence, habitat use in the most recently burnt site (< 1 year post-fire) is statistically different from that at medium and older aged sites (5-8 and 30-31 years post-fire) and riparian and edge habitats are preferred across fire ages. Mean arthropod energy content increased significantly between fire ages < 1 and 16, corresponding to the lowest and highest arthropod availabilities, respectively, but not to habitat use *per se*. Considering the discrepancy in the years of sampling and fire ages for the habitat use and arthropod prey studies, it is difficult to make any definitive statements regarding their relationship and possible interactions with fire age. Although previous such research is very limited, a number of studies have noted that fire effects on food resources influence avian habitat use and subsequent patterns in species richness and abundance (Jordan 1987a; Woinarski 1990; Stuart-Smith *et al.* 2002; Loyn *et al.* 2003; Ward and Paton 2004b).

Decisions by birds at the home range scale may result from an assessment of costs and benefits (e.g. food availability vs. predation risk) as propounded by optimal foraging theory regarding patch use (Hutto 1985; Begon *et al.* 1996), and also may be reflected in patterns at coarser scales (Wiens 1985). Although not investigated in this study, Maguire's (2006b) study on the Southern Emu-wren in Victoria showed that these decisions have fitness benefits, in that pairs that held territories with a greater proportion of habitats that supported the highest abundance of preferred prey orders throughout the year (i.e. tall shrublands, closed heathlands and sedge/rushlands) tended to produce larger clutches and/or fledged more offspring. Factors such as fire may also influence fitness, for example recently burnt sites may not provide adequate food or nesting resources and result in delayed breeding and lower survival rates of offspring (Brooker and Rowley 1991; Russell and Rowley 1998). Edaphic factors may likewise influence fitness. For example, Ormerod *et al.*'s (1991)

results, although on Dippers (*Cinclus cinclus*) which are aquatic foragers, indicated that high soil acidity may have adversely affected breeding performance and nestling growth by reducing the availability of invertebrate prey resources. The relative differences in soil fertilities between the Lake St Clair and Lake Pedder may similarly influence prey availability and subsequently impact fitness. However, the extent to which such factors influence fitness may also depend on what resources are actually available to individuals, particularly in the case of territorial passerines.

Limitations

Although use versus availability studies are common throughout the wildlife literature, Garshelis (2000) identified two ‘fatal flaws’ in such studies: 1) in the absence of preference habitats are used proportionate to availability, and 2) selection of habitats is directly related to those habitats conferring fitness benefits to the animal. The functional relationship between area and use is likely variable, and may be inconsequential in relation to the specific resource requirements of an individual and the relevant habitat attributes (Garshelis 2000). In addition, individuals may not have free and equal access to all habitats as defined (Garshelis 2000) due to intra- and interspecific interactions, especially regarding territoriality (Thomas and Taylor 1990; Alldredge and Ratti 1992). The spatiotemporal scales at which availability is both defined and measured relative to use may have a significant bearing on the results (Johnson 1980; Dodge *et al.* 1990; McClean *et al.* 1998). Furthermore, due to the unit sum constraint the proportions of different habitats are not independent, and thus if one habitat has a low proportional use others will have a correspondingly high use (Johnson 1980; Aebischer *et al.* 1993), whereas in reality individuals may be selecting for a mosaic of habitats rather than one patch over another to meet different resource requirements (Garshelis 2000). Furthermore, presence in a habitat is not necessarily positively correlated with habitat quality (Van Horne 1983). Other factors that may influence habitat selection and subsequent observations include predator avoidance (Maguire 2006a, 2006b), differences in physiology, morphology, behaviour, and resource requirements (Morris 1987), age and sex (Block and Brennan 1993), and sampling biases such as detectability (Manly 2002) and responsive movement (Buckland *et al.* 2001). Patterns are reflective of the summation of individual choices (processes) within home ranges; however, extrinsic factors operating at larger scales may influence these processes as well (Wiens 1985). Stochastic events such as fires may alter habitat and affect subsequent selection from the microhabitat scale to the geographic scale and from the immediate to the long term. Due to the number of assumptions that have to be made for all habitat use studies and the specific limitations for this study (i.e. only one year of habitat use data), all inferences made are in relation to ‘averages’ during the course of the study and for the populations and study area as defined (Alldredge and Ratti 1992; Block and Brennan 1993; Manly *et al.* 2002). Thus, the selection

probabilities are still considered to be valid but the associated errors may not reflect the actual variation in the populations being studied (Manly *et al.* 2002).

The primary issues with avian habitat use and food resource studies are biased sampling of food resources and definitions of availability in both space and time (Block and Brennan 1993). As no sampling methods are considered to be capable of collecting all individuals from all orders present in a plot (Southwood and Henderson 2000), these results may have underestimated actual abundance and diversity. For example, during the course of vacuum sampling two large orthopterans (~ 3 cm) and one large dipteran (~ 4 cm) were observed within or adjacent to the quadrat but were able to escape collection, although such apparent evasive action was expected (Ausden 1996). Overall extraction efficiency of vacuum sampling under some conditions has been shown to be reduced in taller vegetation (~ > 50 cm; De Barro 1991; Brodmann and Reyer 1999; Hossain *et al.* 1999); therefore, differences between riparian and matrix habitats may be greater than indicated by these results. It is also important to note that these are measures of standing crop within matrix and riparian habitats and not actual productivity, and that these figures essentially represent a snap-shot in both space (i.e. 11 sites at Lake St Clair) and time (i.e. 4 days of sampling) and do not encompass the spatial, seasonal, and annual variation that would be expected in arthropod communities (Wiens 1984; Hutto 1990; Driessen and Greenslade 2004). For example, overall arthropod catches in buttongrass moorlands are known to be significantly lower in winter (M. Driessen unpublished data). Finally, measures of abundance and diversity may be confounded with changes in arthropod activity, particularly in relation to post-fire succession (M. Driessen pers. comm. 2007), although this is likely to be more of an issue with passive sampling methods such as pitfall traps.

Although the prey orders included in this study were based on species-specific dietary data from the literature, there are a range of factors that may influence which arthropods are actually consumed by the avifauna. There can be a complex interaction between interspecific foraging methods and arthropod size, conspicuousness, activity patterns, mobility, abundance, microhabitats, and weather (Bryant 1973; Hutto 1980; Hutto 1981; Cooper and Whitmore 1990; Wolda 1990). Some studies suggest that foraging choices are ultimately based on profitability, particularly during the breeding season, such that birds will pursue large and/or abundant prey that are easily captured in order to maximize net energy gain per unit time (Bryant 1973; Brodmann and Reyer 1999). Such important patterns may be masked by only identifying arthropods to the ordinal level, as there is a risk that nonprey species may actually be driving the observed patterns in arthropod abundance and energy content and obscure any relationships with avian habitat use (Wolda 1990). The scale of measurement may also influence results if the sampling of prey resources is not

commensurate with that of foraging (Hutto 1990). The extent to which seasonal differences may affect arthropods is unknown, although community composition and energy content are known to vary with seasons (Norberg 1978; Driessen and Greenslade 2004; M. Driessen unpublished data; pers. obs.), which may, in turn, influence the relative use of these habitats (Hutto 1980). Lastly, birds may directly and indirectly affect insect populations (Hutto 1985; Otvos 1979 cited in Block and Brennan 1993); thus, sampled areas may represent depleted patches. When only measuring the standing crop, it must be assumed that values are representative of the resources in that microhabitat both before and after the sampling event (Hutto 1990). Ultimately, one must decide whether the chosen methods produce reliable results regarding the use and actual availability of habitat and food resources (Hutto 1990). Despite some of these limitations, and particularly the fact that the arthropod surveys were not conducted synchronously with the habitat use surveys, it is believed that these results provided a reasonable first approximation of the relative availability of potential arthropod prey resources between matrix and riparian habitats in buttongrass moorlands. These results may help to explain, in part, the observed patterns in avian habitat selection in moorlands.

The implications of the results from this study for avifaunal conservation and fire management of Tasmanian buttongrass moorlands are summarised in Chapter 7.

Short-term avifaunal composition and densities in relation to hazard-reduction burning of buttongrass moorlands in the Tasmanian Wilderness World Heritage Area

Introduction

Fire has been part of the Tasmanian environment for millions of years and was probably used as a land management tool by Tasmanian Aborigines since approximately 40,000-70,000 years BP (Plomley 1966; Kirkpatrick *et al.* 1978; Kee *et al.* 1993; Jackson 1999a; Kershaw *et al.* 2002; Gammage 2008; see Chapter 1). Historical records indicate that Tasmanian Aborigines actively burnt buttongrass moorlands throughout western Tasmania at the time of early-European settlement in the early 1800s (Plomley 1966; Gammage 2008). It has been speculated that Aborigines probably lit recurrent fires (e.g. inter-fire interval ≤ 20 years) under weather conditions that would have resulted in low-intensity burns that were largely restricted to moorland vegetation (Marsden-Smedley 1998a). Early European use of fire was variable, but showed a notable departure from Aboriginal-style burning practices, resulting in a number of high-intensity, landscape-scale fires that burnt large areas of both moorland and fire-sensitive vegetation (e.g. rainforest and alpine communities) (Marsden-Smedley 1998b; Johnson and Marsden-Smedley 2002; Pyrke and Marsden-Smedley 2005).

Currently, most fire management activities in buttongrass moorlands are focused on resource protection and consist of limited wildfire suppression and frequent tactical hazard-reduction burning (i.e. inter-fire interval ~ 5 -8 years) in areas that are at a high risk of accidental and arson ignitions, such as the medium productivity sites along the Lyell Highway near Lake St Clair in, and adjacent to, the Tasmanian Wilderness World Heritage Area (TWWHA) (PWS 1996; Marsden-Smedley *et al.* 1999; Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley *et al.* 2001; Marsden-Smedley 2009; PWS unpublished data; see Chapter 2). The primary objective of hazard-reduction burning in buttongrass moorlands is to reduce $> 70\%$ of the fuel load across $> 70\%$ of the site being burnt (PWS 1996; Marsden-Smedley *et al.* 1999). Reduced fuel loads help ensure that subsequent fires in intervening years will have lower rates of spread and shorter flame heights, and thus increase the likelihood of successful suppression (Marsden-Smedley and Catchpole 1995b; Marsden-Smedley *et al.* 1999). It is estimated that less than 2% of moorlands throughout Tasmania is currently subjected to tactical hazard-reduction burning (Driessen 2006). Although limited in scope, it is recognised that high-frequency fire regimes may adversely affect biodiversity and cause long-term community changes in the targeted areas (Jackson 1978; Marsden-Smedley and

Kirkpatrick 2000; Pyrke and Marsden-Smedley 2005). Thus, fire management is arguably the primary deterministic factor affecting Tasmanian buttongrass moorland ecosystems, particularly in the TWWHA, where they are largely protected from other potential anthropogenic threats.

Proposed future fire management strategies include an overall increase in the total area of moorlands burnt on an annual basis (i.e. 5-10%), using a combination of broad-scale ecosystem management burning and tactical hazard-reduction burning in high risk areas, as well as wildfire suppression when and where appropriate (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley *et al.* 2001; PWS 2004; King *et al.* 2006, 2008; Marsden-Smedley 2009). It is thought that such a strategy would mimic Aboriginal burning regimes and help to conserve fire-adapted ecosystems (e.g. moorlands, wet sclerophyll woodlands and forests), while limiting the incidence, extent, and severity of unplanned fires and their adverse impacts on fire-sensitive resources (e.g. peat, temperate rainforests, alpine communities) and other assets (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley *et al.* 2001; Pyrke and Marsden-Smedley 2005; PWS 2004; King *et al.* 2006, 2008; J. Marsden-Smedley pers. comm. 2007). Treatment levels at the lower end of this range (i.e. ~ 5%) would represent a fairly modest increase on current levels, but at the upper end (i.e. ~ 10%) would result in higher fire frequencies than are believed to have characterised Aboriginal and early-European fire regimes, and may adversely impact moorland ecosystems (Marsden-Smedley 1998b; Marsden-Smedley and Kirkpatrick 2000; PWS 2004; Pyrke and Marsden-Smedley 2005; King *et al.* 2006, 2008). Accordingly, both current fire management practices and proposed future strategies have generated vigorous debate over the potential short- to long-term effects of fire on biogeodiversity, including the moorland avifauna, and a consensus on appropriate fire management both within and outside of the TWWHA remains elusive (DPIW 2007; May and Balmer 2008; see Chapter 7).

Although many birds are probably able to survive low- to moderate-intensity fires typical of hazard-reduction burns in buttongrass moorlands (Cowley *et al.* 1969; Recher and Christensen 1981; Rowley and Brooker 1987; Wooller and Calver 1988; Marsden-Smedley and Catchpole 1995c; Marsden-Smedley *et al.* 1999), their ability to persist and reproduce in the post-fire environment is less certain and may be influenced by a wide range of species-, site-, and fire-specific factors (Brooker and Rowley 1991; Keith *et al.* 2002a; Whelan *et al.* 2002; Bradstock *et al.* 2005; see Chapters 1, 4-5). Despite the pyrogenic nature of buttongrass moorlands (Pyrke and Marsden-Smedley 2005), some fire-sensitive attributes are exhibited by a number of the species that comprise the moorland avifauna, including the resident Southern Emu-wren, Striated Fieldwren, and Ground Parrot (Meredith *et al.* 1984; Bryant 1990; Gosper and Baker 1997; Pickett 2000; Higgins and Peter 2002; Wilson and

Paton 2004; Tasker and Baker 2005; see Chapter 4). Inappropriate fire regimes are a demonstrated or speculated threat to mainland populations (Garnett 1992; Brown *et al.* 1993; Garnett and Crowley 2000; see Chapters 1 and 4) and the space-for-time (SFT) studies presented in Chapters 4 and 5 highlight some of the potential effects of fire on the resident and non-resident species of Tasmanian moorlands over successional time scales (i.e. up to ~ 50 years post-fire). However, the inferences that can be drawn from SFT studies are somewhat limited due to the possibility of inter-site variability in abiotic and biotic factors confounding or compounding observed differences in populations from any effects of fire *per se* (Whelan 1995; Loyn 1999; Woinarski 1999b; Block *et al.* 2001; see Chapters 2 and 4). An alternative study design is before-after-control-impact (BACI) (Stewart-Oaten *et al.* 1986), that may include paired impact and control sites that are surveyed before and after the impact (i.e. fire) (Smith 2002). This provides a basis for stronger inferences since any observed changes in community composition and abundance can more reasonably be attributed to the effects of fire *per se* (Whelan 1995; Loyn 1999; Parker and Wiens 2005).

During the autumn and spring of 2005, the Parks and Wildlife Service (PWS) conducted hazard-reduction burns in the Lake St Clair area, including moderate productivity, eastern moorlands at Travellers Rest (TRR, last burnt 1973) and Navarre Plains East (NPE, 1988), respectively (see Chapter 2). Since these sites were surveyed during three seasons in 2004, along with both Beehive Canal North (BCN, 1974) and Bedlam (BED, 1990) which remained unburnt, a valuable opportunity was available to more rigorously assess the short-term impacts of fire on the moorland avifauna within a paired BACI framework. Such opportunistic studies conducted in other regions of Australia have provided important guidance for avian conservation and fire management (e.g. Brooker and Rowley 1991; Recher 1997; Loyn 1997). In this study, distance sampling and variable circular-plots were used within a paired BACI design to quantify pre- and post-fire (i.e. before- and after-impact) avifaunal diversity and density in paired unburnt (i.e. control during time of study) and burnt (i.e. impact) medium productivity eastern moorlands at Lake St Clair, Tasmania. The primary aims of this study were 1) to investigate the short-term impacts (5-18 months post-fire) of low intensity hazard-reduction burns on avifaunal diversity and density, with a focus on the Southern Emu-wren and Striated Fieldwren; 2) to assess the influence of site- and fire-specific factors on species response patterns; and 3) to provide information to help guide fire management and conservation of the buttongrass moorland avifauna.

Methods

Study area

Details of the site descriptions, fire histories, vegetation configurations, and floristics of the study sites are provided in Chapters 2 and 3. This study was limited to four sites (i.e. BED, NPE, BCN, TRR) on the moderate-productivity doleritic geologies at Lake St Clair, Tasmania. The moorland matrix primarily consists of Eastern Buttongrass Moorland with a small area of Subalpine Sedgeland at BCN (for vegetation classes see Table 4, Chapter 3). The matrix is scattered with Eastern Wood Copses, interspersed and bordered by typically small perennial watercourses (i.e. first - third order streams; after Strahler 1952) forming Eastern Buttongrass Riparian Zones, and primarily bordered by *Eucalyptus delegatensis* Woodlands and roadside vegetation (i.e. Cleared or Disturbed Land), all of which form relatively distinct edges with the moorland matrix.

Hazard-reduction burns were conducted by PWS on 21 April 2005 at TRR and 21 September 2005 at NPE, thus eliminating the risk of pseudoreplication among treatment sites (*sensu* Hurlbert 1984). At the time of the hazard-reduction burns the impact sites, NPE and TRR, were approximately 18 and 33 years post-fire, respectively. The control sites, BED and BCN, were chosen since they were the same approximate age as the impact sites at the time of the hazard-reduction burns (i.e. 16 and 32 years post-fire, respectively; see Chapter 2). The burn at NPE was ignited by both an aerial 'Dragon' driptorch and by hand driptorches (ignition pattern not reported). It was of a low intensity (i.e. rate of spread $< 4.5 \text{ m min}^{-1}$ and flame heights $< 4.5 \text{ m}$), burnt approximately 94% of the site as defined in this study, and covered a total of 612 ha, including extensive moorlands to the east and south of the site. The burn at TRR was ignited by hand driptorches (ignition pattern not reported). It was of a low intensity, burnt approximately 93% of the site as defined, and covered a total of 42 ha, including small areas adjacent to the site (Marsden-Smedley and Catchpole 1995c; Marsden-Smedley *et al.* 1999; J. Marsden-Smedley pers. comm. 2007; PWS unpublished data; this study). The hazard-reduction burn objectives were met at both sites, with an estimated average of 70% of aerial fuels burnt over 70% of the areas (PWS 1996; Marsden-Smedley *et al.* 1999; PWS unpublished data; pers. obs.).

Avian survey methods

Survey methods consisted of point and line transect distance surveys conducted during the day for the avifauna as a whole and variable circular plot (VCP) surveys conducted at dusk for the Ground Parrot, as detailed for the SFT surveys in Chapter 4. Transects at each of the four sites were re-established based on the GPS coordinates from the survey points used during the pre-fire SFT surveys in 2004. The post-fire sampling was conducted during the summer (3-6 February), winter (13-17 May), and spring (6-16 October) of 2006; therefore,

each site was surveyed three times and the burn sites (i.e. NPE and TRR) were 5-13 months and 10-18 months post-fire, respectively, during the survey period. Thus, the study design consisted of paired control and burn sites for two fire age classes and at the time of the burns all four sites (i.e. BED-NPE, 16-18 years post-fire; and BCN-TRR, 32-33 years post-fire; see Chapter 2) were considered to be mature moorlands in relation to vegetative cover and other metrics of successional development (Marsden-Smedley and Catchpole 1995b; Marsden-Smedley 1998a; see Chapters 1 and 3). A survey schedule was developed in which the time of day and the order of the control and burn sites were systematically rotated within and across seasons to minimise temporal biases (Mac Nally 1996a). Since the Ground Parrot was absent from the four sites during all of the pre-fire and post-fire VCP surveys, it was excluded from further consideration in this study.

Weather conditions during the surveys were fairly typical for the study area (see Chapter 2); temperature and rainfall data for 2004-2006, when compared to historical climatic data (1961-1990), indicated that there were no major temperature anomalies, while rainfall was 14% below normal during 2006 (BOM 2007). Weather conditions for the pre-fire surveys are detailed in Chapter 4. For the post-fire surveys the mean temperature was 8.7°C (range 1-17°C) during the daytime surveys and 8.7°C (range 3-15°C) at dusk, while mean relative humidity was 76.1% (range 54-100%) during the daytime and 76.3% (range 67-98%) at dusk. During both daytime and dusk surveys cloud cover and wind speeds were highly variable, but survey conditions were typically partly cloudy to overcast (83%; $n = 24$) with calm to light winds (96%) and no precipitation (75%). As reported in Chapter 4, it is unlikely that such weather conditions had a notable influence on the overall results.

Analyses

The resident and non-resident species included in this study and the methods for determining species composition and densities of the control and burn sites during the pre-fire surveys are outlined for the SFT study in Chapter 4. As for the pre-fire surveys, both auditory and visual detections from the point and line transects were used to determine total species presence for the four control and burn sites. Exact distance data (i.e. visual detections) for the post-fire control and burn sites were analysed using Program Distance 5.0 to estimate species densities (Thomas *et al.* 2010) following the methods detailed by Buckland *et al.* (2001, 2004) and Thomas *et al.* (2006, 2010). Separate sets of analyses were conducted for exact point and exact line surveys. Due to the small sample sizes from the post-fire surveys, probabilities (\hat{P}_a) and the standard error (\pm SE) of \hat{P}_a of the number of clusters detected within the surveyed areas (a) (using the associated truncation widths (w) and expected cluster sizes (s)) estimated for the Southern Emu-wren, Striated Fieldwren, and non-resident group for the point and line transects in the SFT study were used as multipliers with a

uniform key function and no adjustment terms to estimate mean stratum-level densities (\hat{D} , birds 10^{-1}) (pooled across the three seasons) for the post-fire control and burn sites. Data were insufficient to obtain reliable results from multiple covariate distance sampling (MCDS) analysis in order to determine whether the treatment covariate influenced detectability for the post-fire surveys. However, based on the MCDS model selection process from the larger pre-fire SFT dataset, the recent-burn covariate was only selected in the case of the Striated Fieldwren and indicated that it had a higher probability of detection in recently burnt sites compared to older sites (i.e. ≤ 1 or > 1 year post-fire at Lake St Clair, respectively). Accordingly, the corresponding covariate-specific probabilities of detection for the pre-fire sites were used to estimate stratum-specific densities for the Striated Fieldwren at the post-fire control and burn sites.

The estimated species-specific mean densities (across seasons) derived from the point and line methods were averaged for each site (Buckland *et al.* 2001; see Chapter 4). The variance of density estimates by species and site was approximated using the delta method and included variance from the encounter rate from the post-fire surveys, and detection probability and cluster size estimation from the pre-fire surveys (Buckland *et al.* 2001; see Chapter 4). Estimates of encounter rate variance were based on a Poisson distribution and each point and line was treated as a replicate (Buckland *et al.* 2001; Thomas *et al.* 2002). The overdispersion parameters (b ; aka variance inflation factor) calculated for each set of analyses in the SFT study following the method described by Buckland *et al.* (2001) were used in order to provide more reliable variance estimates for all of the sites (Lebreton *et al.* 1992; Burnham and Anderson 2002; Franklin *et al.* 2002) and ensure comparability between the pre- and post-fire estimates. Estimates of variance for the final combined point and line density estimates for each species by site were derived following the general methods of Steel and Torrie (1980).

A range of different methods for analysing BACI and related study designs have been extensively discussed and debated in the literature (Hurlbert 1984; Stewart-Oaten *et al.* 1986; Underwood 1994; Conquest 2000; McDonald *et al.* 2000; Murtaugh 2000, 2002, 2003; Smith 2002; Stewart-Oaten 2003; Clarke *et al.* 2006; Buckland *et al.* 2009). In order to address some of the inherent limitations of BACI designs and meet the underlying assumptions of many of these proposed methods, it is necessary to have statistically independent estimates based on large sample sizes, adequate replication within and between sites, and numerous pre- and post-impact survey periods. Murtaugh (2002, 2003, 2007) argued that, even under such ideal circumstances, it may be more appropriate to base inferences on informed interpretation of graphical representations and overall weight of evidence than to solely rely upon results from statistical tests. He demonstrated that

ecological BACI analyses often have high Type I error rates due to serial correlation, and have limited power when more complex models are used to try and correct for this issue. Most importantly, he stated that BACI analyses are based on the untestable assumption that, in the absence of the impact, the mean post-impact difference in response between control and impact sites would have been identical to the mean pre-impact difference (i.e. parallel trajectories over time). Due to these general issues with BACI analyses, as well as the opportunistic nature of this study, the limited dataset, and a lack of statistical independence between sites and survey periods largely arising from the use of shared detection functions (Buckland *et al.* 2001, 2008, 2009), no formal statistical hypothesis tests were used to evaluate the potential influence of the burn treatments on the avifauna.

In order to evaluate short-term responses of moorland avifauna to the hazard-reduction burns, changes in mean species-specific density of the paired replicate sites were estimated by subtracting control density from burn density for the pre-fire (\hat{D} -pre) and post-fire (\hat{D} -post) periods. The standardised species-specific responses to treatments (i.e. difference of differences, or effect size) were estimated by subtracting pre-fire densities from post-fire densities (i.e. \hat{D} -post – \hat{D} -pre = $\hat{\Delta}$) (Stewart-Oaten *et al.* 1986; McCune *et al.* 2002; Hurteau *et al.* 2007; Dickson *et al.* 2009). This approach, based on both audio and visual cues, was also used to assess the overall change in species richness in relation to the burn treatment. Since global detection functions were used to estimate species densities by site, it is important to note that the estimated changes in densities were not independent. Although Buckland *et al.* (2001) present a modified delta method to calculate the associated variance estimates when there is a shared detection function, it is not directly applicable since densities for this study were estimated from both point and line surveys, and were therefore based on two distinct detection functions. Accordingly, estimates of variance for the changes in densities were derived following the general methods of Steel and Torrie (1980), with the caveat that they may be biased and should be considered with caution.

The influence of avifauna community composition and densities on observed patterns in relation to control and burn treatments was explored using non-metric Multidimensional Scaling (MDS) in Primer 5.2.2 (Primer-E Ltd. 2001). The similarities of mean densities contributed by species among all sites both pre- and post-fire ($n = 8$) were calculated using the Bray-Curtis similarity measure since it ignores joint absences and is often recommended for community data (Quinn and Keough 2002; McCune *et al.* 2002). The data were untransformed as there was no interest in reducing the influence of the resident species on the overall results (Clarke and Gorley 2001). The similarity matrix was analysed with MDS using two dimensions with 30 random restarts (Clarke and Warwick 1994). The ordination plot represented apparent patterns between and among control and burn replicate sites during

the pre- and post-fire periods with the underlying similarities in community composition and densities.

Results

Avifaunal responses to hazard-reduction burning

A total of only 20 bird species, in addition to unspecified raptor species, was recorded by audio and/or visual cues during the course of both pre- and post-fire surveys at the four control and burn sites included in this study. Of the 25 species listed for the SFT study in Chapter 4, the only species not detected were the Ground Parrot (*Pezoporus wallicus*), Grey Fantail (*Rhipidura fuliginosa*), Olive Whistler (*Pachycephala olivacea*), Pink Robin (*Petroica rodinogaster*), Strong-billed Honeyeater (*Melithreptus validirostris*), and White-throated Needletail (*Hirundapus caudacutus*), none of which were recorded at any of these sites either pre- or post-fire. Latham's Snipe (*Gallinago hardwickii*) was the only species recorded that was not detected in the SFT study. Three species were only detected in unburnt sites (i.e. pre-fire control and burn and post-fire control): the Beautiful Firetail (*Stagonopleura bella*), Black-headed Honeyeater (*Melithreptus affinis*), and Grey Shrike-thrush (*Colluricincla harmonica*), while two species were only detected at post-fire burn sites, namely Latham's Snipe (*Gallinago hardwickii*) and Black Currawong (*Strepera fuliginosa*). Based on audio and visual cues, there was an overall mean increase of three species in the burn treatments when compared to the controls from the pre- to post-fire periods. Data on the 12 resident and non-resident species included in the following analyses (Table 1) were limited to observations of singles or clusters of birds that were confirmed by visual cues and measured using exact distances from the point and/or line transects ($n = 168$ within w , see Chapter 4).

A summary of estimated mean species densities (across seasons) for the pre- and post-fire control and burn replicate sites is presented in Table 1, and the standardised differences of mean densities (across replicates) between pre- and post-fire estimates for the burn treatments relative to the controls are presented in Figure 1. Overall, the resident Striated Fieldwren and Southern Emu-wren decreased in post-fire densities at burn sites relative to controls, with mean differences of -0.68 ± 0.97 and -1.25 ± 2.96 birds 10 ha^{-1} , respectively. In contrast, all of the non-resident species, except for the Green Rosella, increased in densities relative to controls, with a mean difference across all non-residents of 1.64 ± 0.82 birds 10 ha^{-1} . Of these, the Flame Robin (0.58 ± 0.27 birds 10 ha^{-1}), Superb Fairy-wren (0.56 ± 0.43 birds 10 ha^{-1}), and Richard's Pipit (0.44 ± 0.18 birds 10 ha^{-1}) demonstrated the greatest relative increases in estimated densities.

The large standard errors were due to a number of factors (Table 1, Figure 1), including the small sample sizes, high variability in intra- and inter-site estimated densities, and the propagation of errors due to the series of calculations required to estimate the standardised differences in pre- and post-fire densities (i.e. $\hat{D}_{\text{post}} - \hat{D}_{\text{pre}} = \hat{\Delta}$). Since the confidence intervals included 0 for 7 out of the 12 species, including the two resident species, the data were insufficient to demonstrate a clear treatment effect. For some species the sign and magnitude of standardised differences in estimated densities differed between the paired replicate control-burn sites (i.e. BED-NPE and BCN-TRR). For example, although the mean difference across replicates for the Southern Emu-wren indicated an overall decrease in density due to a relatively large post-fire decrease for BED-NPE (-3.19 ± 5.10 birds 10 ha^{-1}), the difference for BCN-TRR was positive (1.65 ± 3.01 birds 10 ha^{-1}) since it was not detected during the pre-fire surveys but was detected post-fire at TRR (Table 1). There was a similar, although less pronounced, pattern for the Striated Fieldwren with a negative difference for BED-NPE (-1.43 ± 1.49 birds 10 ha^{-1}) and a positive difference for BCN-TRR (0.09 ± 1.25 birds 10 ha^{-1}), since it was likewise not recorded by visual observation during the pre-fire surveys at TRR. However, it was recorded by audio cue in the pre-fire surveys and thus at least utilised, and possibly occupied, the site before the burn (T. Chaudhry unpublished data). Accordingly, the apparent post-fire increase in density of the Striated Fieldwren at TRR may be overestimated (Table 1) and hence the standardised negative difference underestimated (Figure 1). Due to the issues outlined above, these estimates and subsequent inferences should be considered with a high degree of caution.

The non-metric MDS ordination plot of the four control and burn sites pre- and post-fire had a stress value of 0.03 (Figure 2), indicating an excellent two-dimensional ordination that can be interpreted with a high degree of confidence (Clarke and Warwick 1994). Although there are no definitive clusters among the sites, some patterns in avifaunal composition and densities were apparent. Both the control sites (BED and BCN) were more similar in avifaunal composition and densities from the pre- to post-fire periods when compared to the burn sites pre- and post-fire (NPE and TRR), particularly NPE which shows the highest level of intra-site dissimilarity between periods. Although TRR also shows a relatively high level of intra-site dissimilarity, it was apparent that the pre-fire community is the most dissimilar among the pre-fire sites. One possible explanation for this, as noted above, is that the two resident species were not recorded by visual observations during the pre-fire surveys at TRR, but were recorded during the post-fire surveys. Hence, TRR shared greater post-fire similarities with the other sites where these species were likewise recorded. However, since the Striated Fieldwren was not recorded by visual observation in the pre-fire period but was present at the site based on auditory cues, this apparent dissimilarity may be positively biased.

Table 1. Mean density estimates (\hat{D} , birds $10 \text{ ha}^{-1} \pm \text{SE}$) from pre-fire (2004) and post-fire (2006) distance surveys conducted in buttongrass moorland sites ($n = 4$) at Lake St Clair, Tasmania. Refer to Chapter 4 for additional details on methods and species inclusion and order. At the time of the hazard-reduction burns in 2005, BED and NPE were ~ 16 and 18 years post-fire and BCN and TRR were ~ 32 and 33 years post-fire, respectively. At the time of surveys, the burn sites NPE and TRR were 5-13 months and 10-18 months post-fire, respectively. *Denotes moorland residents.

Species	BED		NPE		BCN		TRR	
	Control		Burn		Control		Burn	
	Pre-fire	Post-fire	Pre-fire	Post-fire	Pre-fire	Post-fire	Pre-fire	Post-fire
Striated Fieldwren*								
<i>Calamanthus fuliginosus</i> (CAFU)	0.16 \pm 0.22	1.29 \pm 1.31	0.74 \pm 0.65	0.44 \pm 0.18	0.91 \pm 1.00	0.91 \pm 0.74	0.00 \pm 0.00	0.09 \pm 0.06
Southern Emu-wren*								
<i>Stipiturus malachurus</i> (STMA)	2.02 \pm 3.66	2.76 \pm 2.69	2.45 \pm 2.32	0.00 \pm 0.00	1.64 \pm 2.31	1.23 \pm 1.49	0.00 \pm 0.00	1.24 \pm 1.22
Tasmanian Thornbill								
<i>Acanthiza ewingii</i> (ACEW)	0.00 \pm 0.00	0.22 \pm 0.26	0.00 \pm 0.00	0.13 \pm 0.10	0.52 \pm 0.30	0.27 \pm 0.22	0.32 \pm 0.21	0.81 \pm 0.34
Crescent Honeyeater								
<i>Phylidonyris pyrrhoptera</i> (PHPY)	0.47 \pm 0.37	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.15 \pm 0.16	0.00 \pm 0.00	0.12 \pm 0.13	0.17 \pm 0.14
Superb Fairy-wren								
<i>Malurus cyaneus</i> (MACY)	1.19 \pm 0.61	0.72 \pm 0.47	0.00 \pm 0.00	0.46 \pm 0.21	0.12 \pm 0.14	0.00 \pm 0.00	0.41 \pm 0.23	0.35 \pm 0.23
New Holland Honeyeater								
<i>Phylidonyris novaehollandiae</i> (PHNO)	0.22 \pm 0.26	0.00 \pm 0.00	0.00 \pm 0.00	0.06 \pm 0.06	0.00 \pm 0.00	0.00 \pm 0.00	0.09 \pm 0.10	0.00 \pm 0.00
Richard's Pipit								
<i>Anthus novaeseelandiae</i> (ANNO)	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.32 \pm 0.16	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.67 \pm 0.32
Yellow-throated Honeyeater								
<i>Lichenostomus flavicollis</i> (LIFL)	0.00 \pm 0.00	0.00 \pm 0.00	0.07 \pm 0.08	0.21 \pm 0.14	0.15 \pm 0.16	0.15 \pm 0.16	0.00 \pm 0.00	0.00 \pm 0.00
Green Rosella								
<i>Playcercus caledonicus</i> (PLCA)	0.00 \pm 0.00	0.22 \pm 0.26	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.12 \pm 0.14	0.00 \pm 0.00	0.09 \pm 0.10
Black Currawong								
<i>Strepera fuliginosa</i> (STFU)	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.20 \pm 0.13	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00
Flame Robin								
<i>Petroica phoenicea</i> (PEPH)	0.00 \pm 0.00	0.00 \pm 0.00	0.00 \pm 0.00	0.67 \pm 0.26	0.43 \pm 0.28	0.40 \pm 0.26	0.12 \pm 0.13	0.49 \pm 0.25
Dusky Robin								
<i>Melanodryas vittata</i> (MEVI)	0.00 \pm 0.00	0.22 \pm 0.26	0.00 \pm 0.00	0.36 \pm 0.17	0.27 \pm 0.22	0.43 \pm 0.28	0.00 \pm 0.00	0.20 \pm 0.16

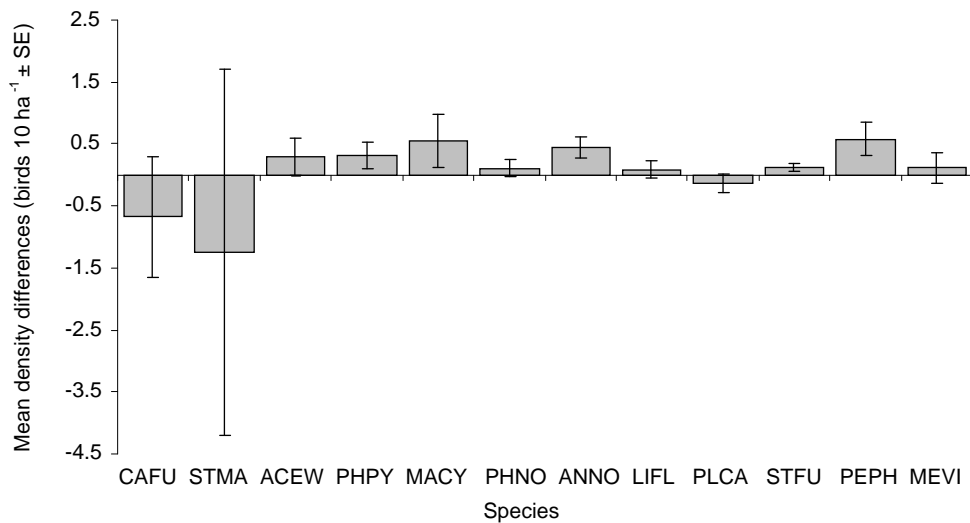


Fig. 1. Standardised differences in estimated pre-fire (2004) and post-fire (2006) mean species densities ($\hat{\Delta}$ birds 10 ha⁻¹ ± SE) across paired control and burn replicate sites ($n = 4$) at Lake St Clair, Tasmania (for species codes see Table 1). Positive and negative density values represent an overall positive or negative response to hazard-reduction burns, respectively (i.e. \hat{D} -post - \hat{D} -pre = $\hat{\Delta}$).

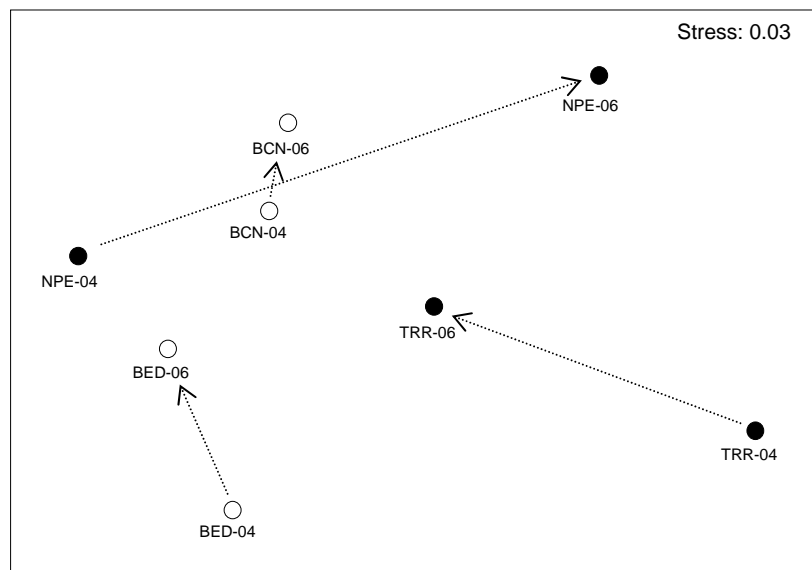


Fig. 2. Two-dimensional non-metric MDS ordination of pre-fire (2004) and post-fire (2006) estimated mean densities (birds 10 ha⁻¹) of 12 bird species at paired control (○) and burn (●) replicate sites (site code-year; → trajectory of change pre- to post-fire) at Lake St Clair, Tasmania (see Table 1).



Fig. 3. A moderate-productivity eastern moorland at Travellers Rest (TRR), Lake St Clair, Tasmania in November 2003, approximately 31 years post-fire.



Fig. 4. TRR in February 2006, 10 months after a low-intensity, hazard-reduction burn conducted by PWS in April 2005. Although vegetative recovery is relatively rapid at such moderate productivity sites, it typically takes at least 4-6 years for cover to return to pre-fire levels (M. Driessen unpublished data; see Chapter 3).



Fig. 5. TRR in February 2006, 10 months after the hazard-reduction burn. A group of three Southern Emu-wrens was observed utilising this burnt matrix habitat during the course of the post-fire survey, despite low levels of cover. Unburnt matrix and riparian vegetation was approximately 100 m from this location within a contiguous patch of buttongrass moorland (see Chapter 2).

Discussion

Short-term avifaunal responses to hazard-reduction burning

This opportunistic BACI study provided insights into the short-term responses of the avifauna to low-intensity hazard-reduction burns in eastern buttongrass moorlands at Lake St Clair, Tasmania. The results from this study suggested that hazard-reduction burning was associated with short-term overall decreases in estimated mean densities of resident species and overall increases in non-resident species in comparison to control sites. The observed patterns reflected the general short-term responses of birds to fire proposed by Pons (2002), including disappearance, persistence, and colonisation. These responses are influenced by both direct (e.g. mortality) and indirect (e.g. habitat modification) fire effects on the avifauna and may vary at the individual level due to factors such as phenotypic plasticity and site tenacity, and at the species level due to autoecological attributes (Pons 2002; Keith *et al.* 2002a; Whelan *et al.* 2002; MacHunter *et al.* 2009).

The observed changes in estimated mean densities of the resident Southern Emu-wren and Striated Fieldwren from the pre-fire to post-fire periods in relation to the burn and control sites were consistent with expectations based on previous research (see Chapters 4-5). Both

species demonstrated short-term overall reductions in estimated mean densities (across seasons and replicates) in relation to hazard-reduction burning during the post-fire period. Previous studies on the Southern Emu-wren and Striated Fieldwren likewise indicated that they were absent, or present in substantially reduced numbers, soon after disturbance by fire and did not start to increase until one to a few years post-fire, although data on the Striated Fieldwren are very limited (Legge 1908; Gellie 1980; Jordan 1987a; McFarland 1988b, 1994; Reilly 1991a; Loyn 1997; MacHunter *et al.* 2009). Specific examples of fire response patterns of the resident species are provided in Chapters 1 and 4. These patterns may be at least partly attributed to their apparent dependence upon adequate cover (for foraging, roosting and nesting), which is often absent or very limited in the post-fire environment (Gellie 1980; Recher 1981; Schodde 1982; Smith 1985; Hopkins and Smith 1996; Woinarski 1999a, 1999b; Gosper and Baker 1997; Wilson and Paton 2004; Maguire 2006a; MacHunter *et al.* 2009). Results presented in Chapter 4 were consistent with these observed patterns, particularly in eastern moorlands at Lake St Clair, where mean densities of both species were relatively low at the youngest site surveyed (i.e. HAR, 1 year post-fire) when compared to mid-seral sites (i.e. 2-16 years post-fire). In addition, results in Chapter 5 indicated that mean abundance and energy content of potential arthropod prey resources were also lower at the youngest sites (i.e. NPE and TRR, < 1 year post-fire) when compared to older sites (i.e. ≥ 3 years post-fire), suggesting limited foraging opportunities for the resident insectivores at recently burnt sites.

Overall differences in densities of the Southern Emu-wren and Striated Fieldwren suggested a negative effect of hazard-reduction burning on these species; however, site-level patterns highlighted the variability in their inferred responses to burning. Specifically, the sign and magnitude of changes for both species differed between the two replicate burn sites (i.e. NPE and TRR) in relation to their respective controls (i.e. BED and BCN), which can be attributed to their autecological attributes and site- and fire-specific context (Keith *et al.* 2002a; Whelan *et al.* 2002; Bradstock *et al.* 2005; MacHunter *et al.* 2009; see Chapter 1). Although both species were present at NPE during the pre-fire period, the Southern Emu-wren was absent and the Striated Fieldwren had reduced densities during the post-fire period. In contrast, the reported increases in densities at TRR were in relation to their apparent absence during the pre-fire surveys. In the case of the Southern Emu-wren, there was no evidence either on- or off-survey to indicate its presence at the site during the pre-fire period, although it is likely they were inhabiting part of the adjacent moorland outside of the surveyed area prior to the fire since it was the closest potential source habitat (~ 35 ha) (Figure 5). In the case of the Striated Fieldwren, as discussed above, auditory observations during the pre-fire surveys indicated that it was present at TRR; however, since density estimates were only based on visual detections (see Chapter 4), the apparent slight increase

in post-fire density may simply be due to a failure to visually detect it during the pre-fire surveys. Overall, the magnitude of post-fire decreases in densities for both residents at NPE was considerably greater relative to their increases at TRR.

Post-fire reductions in densities at NPE relative to TRR may have been caused by direct mortality (i.e. heat or asphyxiation) including loss of clutches, indirect mortality (i.e. loss of territory, starvation, predation), or dispersal to unburnt sites (Fletcher 1913a, 1946; Fox 1978; Gellie 1980; Bigalke and Willan 1984; Keith *et al.* 2002a; Pons 2002; Pickett 2005). Many birds, including both resident species, are able to avoid fire fronts and survive low-intensity fires, such as those reported in this study (Fletcher 1913a; McNamara 1945; Gellie 1980; Schodde 1982; Woinarski and Recher 1997; Burbidge 2003; pers. obs.). However, different ignition methods were used at NPE (i.e. aerial 'Dragon' and hand driptorches) and TRR (i.e. hand driptorches) which may have influenced the likelihood of escape and survivability due to the associated differences in fire behaviour (Parr and Chown 2003). For example, the aerial ignition at NPE would have been more likely to involve line-ignition patterns with multiple fire fronts converging, resulting in higher fire intensities and a lower probability of escape. In addition, since the burn at NPE was conducted in September, during the reported breeding seasons for both resident species (Legge 1908; Dove 1912; Fletcher 1913a; Fletcher 1918; Sharland 1953; Napier 1969), it is possible that clutches were lost, young may have been killed due to their limited mobility, and/or breeding adults may have been killed due to a reluctance to leave nest sites in adequate time to escape (Gellie 1980). In contrast, the burn at TRR was conducted in April, outside of their breeding seasons, and thus would have posed less of a threat to birds that may have colonised the site during the period between the pre- and post-fire surveys. The burn at NPE was also more extensive than the one at TRR (i.e. 612 ha vs. 42 ha), further from unburnt habitat (i.e. ~ 500 m vs. ~ 150 m from the site centroids to nearest habitat > 1 ha), and shared limited edge with adjacent unburnt habitat (i.e. ~ 200 m vs. 1500 m) (Marsden-Smedley *et al.* 1999; J. Marsden-Smedley pers. comm. 2007; PWS unpublished data; this study). Previous research indicates that the likelihood of survival, persistence, and/or (re)colonisation by the residents is dependent upon the availability of adequate fire refugia within or nearby territories due to their dependence on adequate cover and limited flight capabilities (Fletcher 1913a; Cooper 1974; Gellie 1980; Recher and Christensen 1981; Pringle 1982a, 1982b; Emison *et al.* 1987; Britton 2004; Pickett 2005). Results from Chapters 4 and 5 likewise indicated that the persistence of the resident species at the most recently burnt site at Lake St Clair (i.e. HAR, 1 year-post fire) was dependent on the presence of unburnt or partially burnt riparian habitat within and adjacent to the site. Accordingly, these differences in site- and fire-specific factors at NPE and TRR helped to explain the respective differences in post-fire responses of the residents in comparison to the controls.

These observations are also supported by research on the Eastern Bristlebird (*Dasyornis brachypterus*), which is likewise a cryptic, ground-dwelling, insectivorous passerine with limited flight capabilities that inhabits dense sedgeland and heathlands (Baker 1997, 2000, 2002). Recent studies on the Eastern Bristlebird by Bain *et al.* (2008) and Lindenmayer *et al.* (2009) demonstrated that despite predictions based on such fire-sensitive attributes, it can persist in and rapidly recolonise recently burnt habitat, provided that adequate unburnt refugia is within or nearby occupied territories. This further highlights that short-term responses of such species is highly dependent upon spatiotemporal context.

The non-resident species included in this study demonstrated short-term overall increases in estimated mean densities (across seasons and replicates) in relation to hazard-reduction burning during the post-fire period, in contrast to the patterns observed for the two resident species. Although there was some variability in species-specific responses between the replicate sites, the signs of the mean differences between the burn and control sites from the pre- to post-fire periods were the same for all but two species (i.e. Tasmanian Thornbill and New Holland Honeyeater). Of the non-resident species, the Flame Robin showed the greatest positive mean difference at the burn sites relative to the controls from the pre- to post-fire period, followed by the Superb Fairy-wren and Richard's Pipit. Both the Flame Robin and Richard's Pipit are associated with open and early seral habitat and have been shown to colonise and increase in abundance in recently burnt habitats, including buttongrass moorlands in Tasmania and heathlands and associated habitats on the mainland (Loyn 1985, 1997; Ratkowski 1985; Smith 1985; McFarland 1988b; Reilly 1991b; Brown *et al.* 1993; Schulz and Kirstensen 1994; Woinarski and Recher 1997; Watts 2002; Loyn *et al.* 2003; Paton *et al.* 2005; Recher 2005; MacHunter *et al.* 2009). Superb Fairy-wrens have been recorded returning to burnt areas within a few weeks to a few months after fire, although they generally require a mosaic of both open areas and those with adequate cover, as was the case at both of the burn sites (Ratkowsky 1978; Recher 1981; Schodde 1982; Loyn 1997; Rowley and Russell 1997; Taylor *et al.* 1997; Watts 2002; Loyn *et al.* 2003; see Chapter 3). Both the Black Currawong and Dusky Robin have also previously been documented as being common in, or opportunistically utilising, recently burnt moorlands and other open and early seral habitats in Tasmania (Green and Mollison 1961; Gellie 1980; Ratkowski 1984; Brown *et al.* 1993; Schulz and Kirstensen 1994; Taylor *et al.* 1997). None of the above five species was recorded at NPE during the pre-fire period, while the Richard's Pipit and Dusky Robin were also absent at TRR during the pre-fire period. This suggests that these species had colonised, or at least opportunistically utilised, both sites during the post-fire period. They also demonstrated greater use of the burnt matrix habitat at these sites when compared to the riparian and edge habitats, whereas their use of the unburnt matrix at the post-fire control sites was notably less (i.e. 59% vs. 14%, respectively) (This study; see Chapter 5). This may

be in part due to the increased foraging opportunities for seeds and arthropods afforded by the open, burnt ground (Gellie 1980; Brown *et al.* 1993; Higgins *et al.* 2001, 2006; Higgins and Peter 2002; see Chapter 1). Results from the SFT study in Chapter 4 likewise demonstrated that these five species were relatively common in recently burnt eastern moorlands at Lake St Clair and blanket moorlands at Lake Pedder (i.e. 1-3 years post-fire).

In contrast, the other four non-resident species (i.e. Tasmanian Thornbill and Crescent, New Holland, and Yellow-throated Honeyeaters) had increased densities at the post-fire burn sites relative to the controls and are typically absent or occur in reduced numbers in recently burnt habitat due to their preference for dense scrub and forested habitats (Ridpath and Moreau 1966; Thomas 1979; Ratkowsky 1978, 1979, 1985; Recher and Christensen 1981; Reilly 1991b; Loyn 1997; Taylor *et al.* 1997; Watts 2002; MacHunter *et al.* 2009). However, the majority of observations of these species at the post-fire burn sites were in unburnt or partially burnt riparian and edge habitats that share greater structural and floristic similarities with their core habitats compared to the moorland matrix (T. Chaudhry, unpublished data; see Chapter 3). This pattern of habitat use is consistent with results from Chapter 5, which demonstrated that the non-resident species had higher relative selection probabilities for riparian and edge habitats compared to the matrix, especially at the recently burnt site at Lake St Clair (i.e. HAR, 1 year post-fire). Finally, the Green Rosella was the only non-resident species that showed an overall decrease in density at the burn sites, consistent with previous research where it was more commonly recorded in 'unburnt' forested habitats (Ratkowsky 1979, 1985).

Community-level patterns in avian composition and densities varied between the control and burn treatments and from the pre- to post-fire periods. These patterns partially reflected the differing combinations of disappearance, persistence, and colonisation response patterns by the resident and non-resident species at each site. Based on both audio and visual cues, there was a slight overall increase in species richness in relation to the burn treatments, although the particular composition of species varied among the sites. Based on the 12 species included in the multivariate analysis, there were greater similarities in species composition and densities among the control sites from the pre- to post-fire periods when compared to the burn sites, in part due to the influence of early seral coloniser species at the latter. Post-fire community-level patterns at NPE were most dissimilar among all of the sites, probably due to the unique site- and fire-specific factors there when compared to TRR, as discussed above. In contrast, post-fire patterns at TRR shared greater similarities with the other sites in relation to pre-fire patterns, largely due to the presence of the two resident species, which were not detected in the pre-fire period. These results further highlight that site-level patterns of the avifauna are most likely influenced by numerous factors, in addition to those

inferred from the low-intensity hazard-reduction burns *per se*. Previous observations of heathland avifauna likewise indicate that low-intensity burns may have less pronounced influence on community composition and abundance when compared to more intense and extensive wildfires (Burbidge 2003; Burbidge *et al.* 2005).

Limitations

Although before-after-control-impact (BACI) designs can provide a basis for stronger inferences relative to other study designs, it is still difficult to demonstrate that any observed changes are caused by the treatments *per se* (Whelan 1995; Loyn 1999; Murtaugh 2002, 2003; Smith 2002; Parker and Wiens 2005). This is especially true for such opportunistic field studies as this one that often have limited (or no) replication, small sample sizes with high variance, and are conducted over relatively short time frames that may confound the observed patterns in relation to fire with those from natural annual variation and other factors (Wardell-Johnson and Williams 2000; Hurteau *et al.* 2007; Bain *et al.* 2008; Dickson *et al.* 2009; Lindenmayer *et al.* 2009). Furthermore, as discussed above and shown in Chapters 4 and 5, there are numerous other species-, site-, and fire regime-specific factors (e.g. autecology, habitat configuration, food resources, fire history) that appear to influence the moorland avifauna that could not be quantitatively tested in this study due to the limited dataset. Thus, these results may not be indicative of potential responses of the moorland avifauna to burning beyond the specific species, sites, and pre- and post-fire conditions that characterised this study. Despite such limitations, the observed responses were generally consistent with expectations based on the results from previous research and those presented in preceding chapters, and thus add to the body of knowledge regarding the fire ecology of the buttongrass moorland avifauna.

The implications of the results from this study for avifaunal conservation and fire management of Tasmanian buttongrass moorlands are summarised in Chapter 7.

General discussion

Synthesis

Overview

This is the first quantitative, community-level study of the Tasmanian buttongrass moorland avifauna. The primary aim of this thesis was to investigate the ecology of the Tasmanian buttongrass moorland avifauna and the influence of a range of abiotic and biotic factors on diversity and density, particularly in relation to post-fire succession of blanket moorlands at Lake Pedder and eastern moorlands at Lake St Clair. Below, I provide a synthesis of results from this thesis and discuss them in relation to similarly depauperate sedgeland and heathland communities within a national and global context, as well as models of post-fire faunal succession within a theoretical context. I also discuss the implications of these findings for fire and conservation management in Tasmanian moorlands and I conclude this thesis by providing recommendations for future directions.

Since this constitutes the first major study on the buttongrass moorland avifauna in some respects the results should be considered as preliminary. In particular, it should be noted that sample sizes proved to be inadequate to conduct the full range of desired analyses, due to the cryptic nature and low apparent densities of the resident species and the limited time and resources that are characteristic of many doctoral studies. Accordingly, the strength of inferences that I have been able to draw from my dataset are limited to some degree. Nevertheless, the outcomes of this thesis and subsequent publications will hopefully serve as an impetus for continued investigations into this fire-adapted, World Heritage ecosystem that we are just beginning to understand and value.

The depauperate avifaunas of sedgelands and heathlands

During the course of the SFT and BACI surveys conducted for this study ($n = 90$) a total of only 26 bird species were recorded, in addition to unspecified raptor species, and were inclusive of almost half of the 55 species reported in the literature as having been observed in buttongrass moorlands and associated habitats throughout Tasmania (see Chapters 1, 4, and 6). This study provides the first quantitative evidence that the core avian community is composed of only three resident species that depend exclusively on moorlands for breeding, feeding, and other resource needs within the study area. The resident species are the Southern Emu-wren and Striated Fieldwren, diminutive passerines that are nearly ubiquitous in moorlands, and the Ground Parrot, one of only four ground-dwelling parrots in the world,

which is more patchily distributed throughout the study area. Only 14 other species, in addition to unspecified raptors, were detected in more than 5% of all point/line transect surveys for both the SFT and BACI studies (including the 13 species specified in Chapter 4 and the Yellow Wattlebird). Since many of these occurred in relatively low densities and are primarily associated with adjacent ecotonal, scrub, woodland, and forest habitats (Watts 2002), they should be regarded as marginal and/or periodic users of moorlands within the study area. The remaining 10 species were essentially opportunistic observations, and are not species typically expected in moorlands *per se* (see Chapters 4 and 6). Thus, the results from this study were in general concurrence with previous qualitative accounts (e.g. Ridpath and Moreau 1966; Brown *et al.* 1993; see Chapter 1) and demonstrated that the Tasmanian buttongrass moorland avifauna is indeed depauperate, characterised by only three, year-round resident species and approximately 13 non-resident species. However, the actual number of species and community composition likely vary with geography, season, and the precise definitions of ‘moorland’ habitat and ‘characteristic’ species. To put this into context, despite the fact that buttongrass moorlands cover approximately 24% of the TWWHA (i.e. 335,000 ha), the avifauna that characterises moorlands includes less than 14% of the 120 terrestrial bird species that occur in this area (Driessen and Mallick 2003; TVMP 2004).

Buttongrass moorlands share floristic and structural similarities with other sedgeland and heathland communities around the world, which are likewise typically found on relatively infertile substrates, are (co-)dominated by sedges and sclerophyllous woody plants (< 2 m), and contain only limited numbers of tall shrubs and trees (Specht 1979a; Jarman *et al.* 1988a; see Chapters 1 and 3). Although communities such as buttongrass moorlands in Tasmania and heather moors in the British Isles occur in cool, temperate regions, many comparable heathlands are found in ‘Mediterranean’ climate zones, such as some heaths of mainland Australia, fynbos of South Africa, coastal sage scrub of California, and matorral of Chile (Cody 1975; Specht 1979a).

Despite the fact that heathlands are widely distributed throughout Australia, from the temperate southeast to the tropical north, all of their avifaunas are characterised by a small proportion of specialist species that are dependent on this habitat throughout the year (Kikkawa *et al.* 1979; McFarland 1988b). Similar patterns emerge when examining the characteristics of sedgeland and heathland avifaunas globally (Table 1). Although these estimates of species richness are obviously contingent upon survey methods and the exact definitions of species-habitat associations (e.g. ‘specialist’, ‘common’, ‘characteristic’), the consistent pattern is that heathland communities are depauperate in comparison to other

vegetation communities found within the same regions, and typically contain only a few species that exclusively rely upon them.

Table 1. Approximate number of species that are specialists or closely associated with global sedgeland, heathlands, and related habitats.

Habitat	No. of species	Region, Country	References
Heathlands and shrublands	4-6	Coastal NSW, Australia	Recher 1981; Martin and Catterall 2001
Graminoid heathlands	15	Southeast QLD, Australia	Dwyer <i>et al.</i> 1979; McFarland 1988b
Fynbos	6	South Africa	Winterbottom 1966; Cody 1975; Bigalke 1979; Siegfried and Crowe 1983; Huntley 1984
Matorral	12	Chile	Cody 1975
Coastal sage scrub	11-32	California, U.S.A.	Cody 1975; Wirtz <i>et al.</i> 1996
Moorlands	9-14	Great Britain	Gimingham <i>et al.</i> 1979; Gimingham <i>et al.</i> 1981; Buchanan <i>et al.</i> 2006; Pearce-Higgins and Grant 2006

The similarities exhibited by heathland avifaunas around the world appear to be a result of a range of shared ecological attributes, most notable of which is vegetation structure. Cody (1975, 1983) examined patterns in species diversities over Mediterranean habitat gradients by comparing the avifaunas of four continents, namely South Africa, South America, North America, and Europe. He found that heathland communities that largely consisted of low vegetation all supported relatively low species richness and diversity when compared to taller and more complex vegetation types within each region. In fact, the overall differences in species diversity along the habitat gradients within each region were greater than those of structurally similar communities between each region. The positive correlation of species diversity with habitat complexity (e.g. height, cover, number of strata) appears to be an emergent theme in most of the studies that have investigated heathland avifaunas, including those in Australia (e.g. Recher 1969; Bigalke 1979; Kikkawa *et al.* 1979; Recher 1981; Brown *et al.* 1993; Pearce-Higgins and Grant 2006). This pattern not only exhibits itself at coarse scales between broad vegetation communities, but also at finer scales within sedgelands and heathlands themselves. For example, results from this study showed that both resident and non-resident species selected the more structurally complex riparian and edge habitats disproportionately to their availability when compared to the relatively simple moorland matrix at Lake Pedder and Lake St Clair (see Chapter 5).

Considering the oligotrophic nature of sedgeland and heathland soils, overall productivity also appears to limit the number of species that can persist solely in these habitats. The infertile and often acidic soils restrict primary productivity, which is reflected throughout the

trophic levels by the relatively limited food resources and consequently small numbers of resident primary and secondary consumers (Ridpath and Moreau 1966; Bigalke 1979; Kikkawa *et al.* 1979; Recher 1981; Milewski 1983; McFarland 1988b; Driessen 2006; Pearce-Higgins and Grant 2006). However, results from this study indicated that microhabitats, such as riparian zones, appear to serve as important resources for insectivorous species, probably due to the greater availability of arthropod prey (i.e. abundance and energy content) and more suitable foraging substrates (see Chapter 5). In addition, riparian zones and edge habitats are often dominated by robust growth-forms of nectar-producing plant species that may likewise serve to boost total productivity and hence overall carrying capacity of these oligotrophic communities (see Chapters 2-3).

The ecological attributes of heathland habitats and surrounding vegetation communities appear to impart a spatiotemporal structure on the avifauna, reflected by the degree of adaptations to these habitats by a limited number of resident specialist species, and the often seasonal associations of opportunistic members of the regional avifauna (Recher 1969; Bigalke 1979; Dwyer *et al.* 1979). A number of resident heathland species from different continents possess striking similarities in body size, morphology, colouration, diets, foraging behaviour, and narrow habitat preferences that exemplify convergent evolution (Recher 1969; Cody 1975, 1983; Milewski 1983). For example, many heathland residents are small wren-like birds that are cover-dependent and insectivorous, such as the Karoo Prinia (*Prinia maculosa*) that inhabits South African fynbos, very similar to the Southern Emu-wren and Striated Fieldwren (Cody 1975; Bigalke 1979; Kikkawa *et al.* 1979; see Chapter 1). The Striated Fieldwren, as well as the Ground Parrot, are considered to be phylogenetically relict heathland species and the Southern Emu-wren is likely to have radiated into heathlands at a later stage in the evolutionary history of the Australian avifauna (Kikkawa *et al.* 1979; Recher 1981; McFarland 1988b; Martin and Catterall 2001). In addition, many heathlands are also used by often nomadic nectivorous and insectivorous birds, such as the Cape Sugarbird (*Promerops cafer*) that utilises South African fynbos, similar to the Crescent, New Holland, and Yellow-throated Honeyeaters observed in this study (Cody 1975; Bigalke 1979; Kikkawa *et al.* 1979). These opportunistic species primarily rely upon the more complex vegetation communities in the surrounding landscape mosaic, but often exploit heathlands when suitable food resources are readily available (Bell 1966; Bigalke 1979; Dwyer *et al.* 1979; Gimingham *et al.* 1979; Kikkawa *et al.* 1979; Recher 1981; Specht 1994; Keith *et al.* 2002a). Such honeyeater species from the family Meliphagidae may have likewise coevolved with heathland vegetation, in particular with nectar-producing plants (e.g. *Banksia* and *Leptospermum* spp.), but have subsequently developed traits that have enabled them to exhibit more catholic habitat use (Kikkawa *et al.* 1979; Burbidge 2003).

In increasingly developed areas on the Australian mainland, the occurrence of heathland specialists may be severely limited by the availability of adequately-sized patches of suitable habitat (i.e. > 5-500 ha, depending on the species), while fragmentation may favour more generalist species that can take advantage of the resulting mosaic of altered and natural vegetation (Rowley and Russell 1997; Martin and Catterall 2001; Lunney *et al.* 2002). Although the TWWHA is largely protected from the impacts of development, results from this study indicated that the relatively 'natural' patterns of buttongrass moorland distribution across the western Tasmanian landscape may also influence the distribution of species such as the Ground Parrot, which was less likely to occur, and was found in lower densities, in relatively isolated moors that formed high-contrast edges with adjacent forest communities (see Chapter 4).

Thus, sedgeland and heathland communities must be examined within a landscape context, with similarities in guild structure and species richness most likely reflecting broad similarities in climate and vegetation structure, while differences in relative abundance may be due to differing availabilities of suitable microhabitat, food, and breeding resources at finer scales (Dwyer *et al.* 1979; Kikkawa *et al.* 1979; Specht 1979b; Cody 1983). Results from this study revealed a similar pattern, with diversity being dominated by species that opportunistically utilised Tasmanian moorlands, such as those typically associated with the interspersed ecotonal, scrub, woodland, and forest habitats (e.g. Tasmanian Thornbill and Superb Fairy-wren); nomadic honeyeater species, particularly during the summer flowering season (e.g. Crescent Honeyeater and New Holland Honeyeater); and summer migrants (e.g. Tree Martin and Welcome Swallow) (see Chapter 4). Habitat use by non-residents mainly occurred in microhabitats (e.g. riparian zones, copses, edges) that shared structural and floristic characteristics with their primary habitats. In contrast, use of the relatively simple moorland matrix was primarily limited to the three resident species (see Chapter 5). However, at both locations the Southern Emu-wren and Striated Fieldwren demonstrated higher relative selection probabilities for riparian and edge habitats when compared to the matrix. Sites with higher proportions of riparian habitat also supported higher densities of both species, particularly the Southern Emu-wren at Lake St Clair (see Chapters 4-5). These findings are in broad agreement with our long-standing understanding of the importance of vegetation structure at varying spatiotemporal scales in determining the composition and relative abundance of avian communities (e.g. MacArthur 1958; Cody 1981; Gilmore 1985; Block and Brennan 1993).

Tasmanian buttongrass moorlands clearly share a number of important attributes with global sedgelands and heathlands. However, the low species diversity found in Tasmanian moorlands is not only a result of the relatively low structural diversity and productivity

characteristic of such habitats, but also reflects an island avifauna (Ridpath and Moreau 1966). The Tasmanian avifauna is impoverished relative to comparably sized areas on the Australian mainland, in part due to the partial geographic barrier caused by Bass Strait during the current and past inter-glacial periods (Ridpath and Moreau 1966). In addition, although Tasmania is the most southerly point along the East Asian-Australasian Flyway, results from this and other studies indicated that either partially or fully migratorial landbirds do not make a large contribution to the diversity of the local sedgeland and heathland avifauna (Ridpath and Moreau 1966; Chan 2001; Watts 2002; see Chapters 4 and 6). Thus, in some respects the characteristic history, geography, climate, and landscape features of Tasmania have provided a template for a globally unique sedgeland and heathland avifauna.

Models of faunal responses to fire disturbance

The shared structural and floristic attributes of global sedgelands and heathlands also contribute to their high flammability (Specht 1979a); therefore, any discussion of these communities must consider the influence that fire has on ecosystem processes and patterns. Fox (1982, 1990a, 1990b) developed the habitat accommodation model for small mammal secondary succession in Australian coastal heathlands based on the principles that species have specific thresholds for habitat colonisation (e.g. minimum % cover) and exhibit differing competitive abilities through the course of succession. The habitat accommodation model predicts that species will occur at a site once the suitability threshold is met, and reach peak abundances as the habitat becomes optimal (i.e. habitat facilitation). Subsequently, abundances may decline and some species may disappear as the habitat succeeds to an unsuitable condition and reduces their competitive advantage over later colonists (i.e. habitat tolerance). In other words, within some faunal communities species will appear, peak, and then may disappear in a sequence that can be predicted based upon species-specific habitat requirements (e.g. low vs. high cover) and the rate of vegetative recovery (Fretwell and Lucas 1969; Fox 1982, 1990a, 1990b). The attributes of a given fire regime (e.g. intensity, season, frequency) may thus influence these successional processes and patterns in both space and time; therefore, it is the continuum of vegetational changes and associated habitat suitability thresholds that determine the timing of faunal succession, and not time since fire *per se* (Fox 1982, 1990a, 1990b; Baker and Whelan 1994; Monamy and Fox 2000). Subsequent studies ranging from small mammal communities in heathlands to ant communities in forests have served to validate the habitat accommodation model (e.g. Twigg *et al.* 1989; Fox 1990a, 1990b; Monamy and Fox 2000; Fox *et al.* 2003).

Whelan *et al.* (2002) provided a broader and more explicitly process-oriented framework within which to examine post-fire population changes in Australian communities. Keith *et al.* (2002a) subsequently applied this framework to heathland ecosystems. They

conceptualise the processes underlying faunal fire response patterns as the interactions between attributes of a species' life cycle with those of the fire regime (e.g. fire behaviour, season, time since last fire), landscape (e.g. slope, fire refugia), and climate (pre- and post-fire). Specific life cycle attributes include behaviour, micro- and macro-scale habitat associations and resource use (i.e. niche breadth), short- and long-distance dispersal capabilities, susceptibility to competition and predation, and reproductive strategies. In conjunction with the environmental attributes, these will have a bearing on a species' ability to survive the fire event, persist in or opportunistically utilise the post-fire environment, (re)colonise from unburnt areas, and successfully reproduce (Recher and Christensen 1981; Fox 1982; Bigalke and Willan 1984; Whelan 1995; Sutherland and Dickman 1999; Keith *et al.* 2002a; Pons 2002; Whelan *et al.* 2002; Bradstock *et al.* 2005). Thus, faunal responses may range from null responses for some generalist bird species (e.g. Kotliar *et al.* 2002), to temporary increases due to greater post-fire availability of food resources (e.g. Woinarski 1990), to reduction and recovery reflecting successional changes in habitat structure and suitability (e.g. Baker 2000) (Woinarski and Recher 1997; Pons 2002; MacHunter *et al.* 2009). However, these patterns are not universal and even intra-specific responses can be highly variable and complex due to site- and fire-specific factors (Mushinsky and Gibson 1991; Fox 1990a, 1990b; Catling *et al.* 2001; Keith *et al.* 2002a; Whelan *et al.* 2002; Burbidge 2003; Bradstock *et al.* 2005; Paton *et al.* 2005).

In view of the above, a logical question is whether or not the observed patterns in avifaunal responses to fire disturbance in Tasmanian buttongrass moorlands can be explained within the frameworks provided by the habitat accommodation and life cycle process models described above. Although the role of fire in moorland avian community dynamics is complex, a number of patterns in diversity, density, habitat use, and resources have emerged. The results from this study suggested that fire is one of the primary factors that influences the composition, densities, and habitat use of bird species in Tasmanian buttongrass moorlands in both the short-term and over successional time scales (i.e. ~1 to > 50 years post-fire; see Chapters 4-6). In particular, fire age was identified as a significant factor in relation to the densities of the three resident species at Lake Pedder and/or Lake St Clair, and at the community level at both locations. In the short term, the resident Southern Emu-wren and Striated Fieldwren showed overall decreases, and the non-resident species overall increases, in densities in relation to fire at Lake St Clair. Over the longer term, all three resident species demonstrated qualitatively similar, although variable, patterns in post-fire changes in densities within each location. The lowest mean densities of the residents generally occurred within the youngest sites surveyed at both Lake Pedder and Lake St Clair (i.e. 3 and 1 years post-fire, respectively). At Lake Pedder the resident species generally increased in densities across the chronosequence, with the highest mean densities at the

oldest sites surveyed (i.e. 54 years post-fire). At Lake St Clair, the mean densities of the residents peaked between 2-8 years post-fire, after which they all gradually declined to their lowest densities at the oldest sites surveyed (i.e. 30-31 years post-fire). In contrast, there were no clear long-term trends in mean densities of the non-resident species in relation to fire age at either location. Thus, both short- and long-term changes in species densities in relation to time since last fire included a combination of the full range of potential patterns identified in models of faunal responses to fire (e.g. persistence, disappearance, reduction and recovery, colonisation, null). These patterns, in part, reflected the rate of vegetative recovery and relative suitability of different seral stages of post-fire vegetation in relation to the specific autecological attributes of the resident and non-resident species in the study area (see Chapters 1, 3-6). Since these patterns would not have become apparent over shorter chronosequences, these results underscore the need for SFT studies to include sites with fire ages that span periods commensurate with successional time scales.

However, results from this study also highlighted that fire age alone cannot account for the full range of variability observed in avifaunal composition, densities, and habitat use within and between sites and across each chronosequence. Observed patterns were also associated with differences in habitat structure, composition, and configuration, as well as the potential availability of arthropod prey resources. In some cases a single variable appeared to have a strong independent influence on estimated densities, such as the positive relationship observed between Southern Emu-wren densities and the proportion of site-level riparian habitat at Lake St Clair (see Chapter 4). However, results suggested that in most cases observed patterns at both the species and community levels could be attributed to multiple variables and their complex interactions (see Chapters 4-5). Numerous other studies in Australian heathlands have demonstrated that bird species composition and abundance may be affected by fire (e.g. Smith 1987; McFarland 1988b, 1994; Brooker and Rowley 1991; Recher 2005), but do not necessarily follow a simple and predictable successional pathway and may be influenced by a range of factors other than, or in combination with, habitat structure and fire age *per se* (Meredith *et al.* 1984; Woinarski and Recher 1997; Baker 2002; Paton *et al.* 2005).

The identified species- and community-level differences in the avifauna between the blanket moorlands at Lake Pedder and eastern moorlands at Lake St Clair may also be attributed to a range of factors that characterise each location. These include the specific structural, floristic, and post-fire successional patterns and processes that typify blanket moorlands and eastern moorlands (i.e. sedgeland vs. graminoid heathlands, respectively), as well as soil productivity (i.e. low vs. moderate fertility), the rate of post-fire vegetative recovery (i.e. slow vs. rapid), altitude (i.e. low vs. moderate), and food resources (see Chapters 4-5). Thus,

differences in post-fire avifaunal responses between the locations partially reflected the differences identified for blanket and eastern moorlands in the fire management models in current use in the TWWHA, particularly in relation to dissimilarities in site productivity and post-fire recovery rates (Marsden-Smedley *et al.* 1999; Marsden-Smedley 2009; see Chapters 1-2).

In summary, Tasmanian buttongrass moorlands are pyrogenic ecosystems; the flora and fauna have adapted to fire over millenia (Kirkpatrick *et al.* 1978). However, fire regimes have changed dramatically over time: from infrequent lightning-caused fire during pre-human times, to frequent and widespread use of fire by Tasmanian Aborigines from the Late Pleistocene throughout much of the Holocene, to relatively few but high-intensity, landscape-scale fires since European settlement from the 1830s-1930s, to relatively limited planned burns and wildfires in current times (Plomley 1966; Marsden-Smedley 1998a; Marsden-Smedley 1998b; Jackson 1999a; Marsden-Smedley and Kirkpatrick 2000; Johnson and Marsden-Smedley 2002; Kershaw *et al.* 2002; Driessen 2006; Fletcher and Thomas 2007; Gammage 2008; see Chapters 1 and 6). Although from an evolutionary perspective it is uncertain approximately when the three moorland resident species may have first occurred within their current range in western Tasmania, it is clear that sufficient time has passed to subject them to a broad range of anthropogenic fire regimes (Ridpath and Moreau 1966).

Accordingly, a critical question is how the three resident species that exhibit some fire-sensitive attributes (i.e. ground-dwelling, cover-dependent, limited dispersal capabilities) have been able to continue to persist in an ecosystem that is both pyrogenic and has been subjected to dramatically shifting fire regimes. Results from this study and previous research indicated that the three resident species can persist in, or rapidly recolonise, some recently burnt sites and occur across a broad range of fire ages and conditions in Tasmanian moorland habitat, although specific patterns may vary in relation to fire age and other factors (Bryant 1991, 1992; see Chapters 4-6). Accordingly, these species appear to possess a degree of phenotypic plasticity and resilience to disturbance by fire that has enabled them to maintain viable populations over time. These traits have similarly been demonstrated for other heathland bird species in Australia and abroad and reflect long-term evolutionary processes (Ridpath and Moreau 1966; Catling and Newsome 1981; Kikkawa *et al.* 1979; Recher and Christensen 1981; Frost 1984; Brooker and Rowley 1991; Pons 2002; Keith *et al.* 2002b; Recher 2005). Although mainland populations have similarly been subjected to varying fire regimes, such changes since European colonisation have been implicated in the local extinction of some mainland populations, and have been identified to be among the primary threats to extant populations (MLRSERT 1998; Garnett 1992; Woinarski 1999a; Garnett and Crowley 2000; Lunney *et al.* 2000; Barrett *et al.* 2003). In contrast, populations

of all three species appear to be relatively secure in western Tasmania (Bryant 1991; Barrett *et al.* 2003; see Chapter 1).

The reason for this apparent disparity in fire impacts on these species in western Tasmania and the Australian mainland is likely due to the dramatically different landscape context between their respective ranges, particularly in relation to development patterns since European colonisation. Buttongrass moorlands are extensively distributed throughout western Tasmania and research indicates that they dominated the landscape throughout the Holocene (Macphail 1979; Brown 1993; Tye 2002; Bridle *et al.* 2003; Fletcher and Thomas 2007). Although the extent of moorlands may currently be more limited due to a decrease in anthropogenic burning since European colonisation (Gammage 2008), extensive tracts still exist (i.e. 0.55 million ha), the majority of which have been protected within the TWWHA since 1982 (i.e. 335,000 ha, or 63%; see Chapter 1) (Smith and Banks 1993; Balmer *et al.* 2004; TVMP 2004). Despite the fire-sensitive attributes of the three resident species, the extensive mosaic of moorlands would have facilitated survival and recolonisation due to the high probability of source populations being in close proximity to recently burnt habitat. In other words, the extensive historical and current distribution of suitable moorland habitat in western Tasmania has probably facilitated the persistence of viable metapopulations, despite varying fire regimes. In contrast, extensive loss and fragmentation of suitable habitat within the ranges of these species on the mainland have been identified as primary threats to extant populations and have already led to declines and local extinctions in some areas (Napier 1969; Forshaw 1981; Sharland 1981; Schodde 1982; Bryant 1991; Garnett 1992; Rowley and Russell 1997; Martin and Catterall 2001; see Chapters 1 and 4). The resulting small and isolated populations are more vulnerable to direct and indirect adverse impacts from fire due to a lack of adequate refugia during and after fire, and the reduced likelihood of recolonisation by distant source populations. Additional threats to these populations include inbreeding suppression, predation, and sudden population declines due to other stochastic factors (Lord 1927; Meredith 1984a; Jordan 1989; Bryant 1991; Pickett 2000; Maguire and Mulder 2004). Thus, the conservation of the Southern Emu-wren, Striated Fieldwren, and Ground Parrot will not only depend upon implementing fire regimes within the historical range and variability of their habitats (Keane *et al.* 2009), as discussed below, but may also require the possible expansion of conservation areas and restoration of degraded habitat, particularly on the Australian mainland.

Implications for fire and conservation management

In July 2007, the first Buttongrass Moorland Management Workshop was held in Hobart, Tasmania (DPIW 2007). It was attended by scientists and land managers from a broad range of disciplines and affiliated organisations (May and Balmer 2008). There appeared to be a shared desire to develop and implement science-based, practical fire management strategies to help conserve Tasmanian moorlands and protect surrounding assets and natural resources, particularly from severe, landscape-scale wildfires. Although issues associated with property protection are very limited in and adjacent to the TWWHA, there was nevertheless a striking lack of consensus on the precise means to achieve this, even within a primarily ecosystem-management context. Over the past decade we have greatly improved our understanding of moorland ecosystems in general, and the role of fire in particular, as a result of research in key fields such as soil science, botany, fire behaviour and fuel modelling, and faunal ecology (see Chapter 1). We now have some of the key pieces to the puzzle, but this workshop was only the first step in trying to put the pieces together. The interdisciplinary and often contentious nature of fire ecology and management, and the all too real risks associated with making poor decisions, render multi-stakeholder decision-making and implementation a daunting task. A detailed discussion of, and possible recommendations for, addressing such ‘big picture’ issues in Australia is well beyond the scope of this thesis.

Aims

The primary aim of these management recommendations is to help ensure, based on the best available science, that fire management strategies support the conservation of the avifauna in Tasmanian buttongrass moorlands. These general guidelines are focused on maintaining resilient populations of the Southern Emu-wren, Striated Fieldwren, and Ground Parrot throughout their ranges since they are the only year-round residents of buttongrass moorlands, are the only members of the avifauna that are entirely dependent upon moorlands in the study area, and some of their populations on the Australian mainland are listed as being threatened, at least in part, due to inappropriate fire regimes (see Chapters 1 and 4). Furthermore, the populations of the Southern Emu-wren (*S. m. littleri*) and Striated Fieldwren (*C. f. diemenensis*) that were the focus of this study are unique subspecies restricted to western Tasmania and thus warrant special protection (Thomas 1979; Schodde and Mason 1999). Since all of the non-resident species recorded in this study, as well as those identified in the literature, are primarily associated with other habitats, their use of moorlands at various post-fire seral stages is relatively opportunistic (see Chapters 1, 4-6). Thus, based on current information, they do not appear to warrant specific management guidelines within moorlands in order to help conserve their broader populations.

While the literature is rife with studies that discuss fire management in relation to avian conservation on mainland Australia (for a review see Woinarski 1999a, 1999b), it is almost entirely devoid of ones on the Tasmanian avifauna, particularly in buttongrass moorlands. Management recommendations in relation to the moorland avifauna only include Bryant's (1991) regarding the Ground Parrot, and Gellie's (1980) on the avifauna as a whole, although he did not provide any quantitative data to substantiate his recommendations. Additional work has been conducted on the critically endangered Orange-bellied Parrot that forages in moorlands areas during the summer breeding season, but this will not be explicitly discussed herein since it occurs in a relatively small part of Tasmania's moorlands, outside the range of this study (Brown and Wilson 1984; Marsden-Smedley *et al.* 2001; M. Holdsworth pers. comm. 2007). Nevertheless, these recommendations should be considered when planning fire management activities within the Orange-bellied Parrot breeding grounds in the Bathurst and Macquarie Harbour basins since these areas also support populations of the three resident species (see Chapter 1; pers. obs.). Unfortunately, the few specific recommendations that have been made regarding the minimisation of adverse fire effects on the moorland avifauna by Bryant (1991) and Gellie (1980) are not explicitly incorporated in current operational guidelines outlined in the *Tasmanian Wilderness World Heritage Area Tactical Fire Management Plan* (PWS 2004) or *Planned burning in Tasmania: operational guidelines and review of current knowledge* (Marsden-Smedley 2009), and are only briefly touched-upon in other fire plans such as the *Lyell Highway Fire Management Plan* that covers the Lake St Clair region (PWS 1996; see Chapters 1 and 6). However, the *Draft Ecological-management Burning Prescriptions* (PWS 2004; Marsden-Smedley 2009) contain a number of general ecological management guidelines that, if formally adopted and implemented, should facilitate conservation of the moorland avifauna.

Guidelines for fire management are provided below based on the autecological attributes of the resident bird species, results from this study, and recommendations from other references, including those from mainland Australia. It should be noted that since additional research is needed to better understand avifaunal dynamics in moorlands (see Future directions), these should be regarded as preliminary recommendations. It is also recognised that managers will need to balance these recommendations for the avifauna against those for other resources, and that a particular regime will not favour all species and resources at a given space and time (Burbidge 2003). Nevertheless, managers should explicitly consider the avifauna in all phases of planning and implementation (e.g. MacHunter *et al.* 2009). While it is expected that these conservative recommendations will likely benefit many faunal species, including non-resident bird species, it is essential to recognise that any myopic and inflexible approaches towards fire management and conservation may be to the detriment of

other taxa and the ecosystem as a whole (Chladil 1991; Agee 1999; Bradstock *et al.* 1995; Kotliar *et al.* 2002; Burbidge 2003; Kotliar *et al.* 2005).

Fire season and intensity

Prescribed burns conducted during the breeding season may result in direct loss of clutches and nestlings, delayed or unsuccessful breeding attempts, and increased predation, particularly for ground-nesting birds such as the resident species (Fletcher 1913a; Gellie 1980; Rowley and Brooker 1987; Reilly 1991b; Maguire 2005; see Chapters 1 and 6). Since identifying the breeding season of the resident and other moorland species was not an objective of this study, these recommendations are primarily based on previous studies and should be modified if additional information indicates that breeding periods differ from those stated below, particularly in relation to climate change, as well as geography, altitude, and site productivity (e.g. Lake Pedder vs. Lake St Clair). In Tasmania, the Southern Emu-wren is known to breed from August through January (Fletcher 1913a, 1918; pers. obs.), the Striated Fieldwren from as early as July through February (Dove 1912; Napier 1969), and the Ground Parrot from October through February (Bryant 1991). Thus, the most conservative approach is to limit both hazard-reduction and ecological-management burning to March - June. This period is more restrictive than that recommended by Bryant (1991) for the Ground Parrot (i.e. April - September), since the Striated Fieldwren starts breeding earlier in Tasmania (Dove 1912). It is also more conservative than the guidelines for ecological-management burning (i.e. Autumn, mid-April - June; Spring, August - September) and hazard-reduction burning (i.e. Autumn, April - early May; Spring, September - mid-October) (Marsden-Smedley *et al.* 1999; PWS 2004; A. Pyke pers. comm. 2007; Marsden-Smedley 2009). The draft ecological burning guidelines also state that when and where feasible, approximately 75% of ecological burns should be conducted in autumn or winter and approximately 25% in spring (PWS 2004; Marsden-Smedley 2009). Based on previous research that indicated birds in fire-adapted ecosystems demonstrated a degree of resilience to a few prescribed burns during the breeding season (Loyn *et al.* 2003; Knapp *et al.* 2009), these guidelines appear to allow for adequately long prescription windows to meet other resource management objectives. However, if spring burns are necessary, adequate areas of unburnt vegetation (i.e. > 30% of site) should be provided for refugia and to facilitate post-fire dispersal (Gellie 1980; Bryant 1991, 1992; PWS 2004; Maguire 2005). Although most prescribed burns in buttongrass moorlands are of a low intensity (Marsden-Smedley *et al.* 1999; J. Marsden-Smedley pers. comm. 2007), these seasonal guidelines should also help to ensure that fire intensities do not exceed a threshold beyond which adverse direct and indirect effects on the avifauna are highly probable (Bain *et al.* 2008).

Fire extent and patchiness

The specific impacts of extensive wildfires on the Tasmanian moorland avifauna are unclear since most of the sites surveyed for this study were subjected to smaller prescribed burns or much older wildfire events (see Chapter 2), which makes it difficult to draw inferences regarding the impacts of large wildfires in the short to medium term. However, due to the fire-sensitive attributes of the resident bird species, it is reasonable to assume that extensive summer wildfires which consume most of the above ground vegetation may result in adverse effects, including mortality, reductions in reproductive success, and local extinctions (Fox 1978; Recher and Christensen 1981; Bryant 1991; Garnett 1992; Loyn 1997; Garnett and Crowley 2002; Pickett 2005; Burbidge *et al.* 2007).

Proposed future fire management strategies for moorlands include an overall increase in the total area burnt on an annual basis (i.e. 5-10%), using a combination of broad-scale ecosystem management burning and tactical hazard-reduction burning in high risk areas (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley *et al.* 2001; PWS 2004; King *et al.* 2006, 2008; Marsden-Smedley 2009). This is considered necessary since a large proportion of moorland is currently in an old-growth seral stage (i.e. > 35 years post-fire) that poses a high fire risk and low probability of successful control (Marsden-Smedley 1998a, 1998b; Marsden-Smedley *et al.* 1999; PWS 2004; J. Marsden-Smedley pers. comm. 2007; A. Pyrke pers. comm. 2007; Marsden-Smedley 2009). It is thought that such a strategy would mimic Aboriginal burning regimes characterised by more frequent, low-intensity fires and help to conserve fire-adapted ecosystems, while limiting the incidence, extent, and severity of unplanned fires and their adverse impact on fire-sensitive resources and other assets (Marsden-Smedley and Kirkpatrick 2000; Marsden-Smedley *et al.* 2001; Pyrke and Marsden-Smedley 2005; PWS 2004; King *et al.* 2006, 2008; Marsden-Smedley pers. comm. 2007).

However, the risk and scale of both planned and unplanned fires should be evaluated on an ongoing basis to help ensure that adequate areas remain unburnt (e.g. ~ 30-70%; Gellie 1980), allowing for the continuing development and availability of mature and old-growth moorland vegetation (i.e. 15-35 and > 35 years post-fire, respectively) across the landscape (Gellie 1980; McFarland 1988b; Woinarski and Recher 1997; Richards *et al.* 1999; Baker 2000; Marsden-Smedley *et al.* 2001; Ward and Paton 2004a; Buchanan *et al.* 2006). This is particularly important for blanket moorlands in the Lake Pedder area, where old-growth sites appear to support relatively high densities of the resident species (see Chapter 4). The maintenance of such large source areas is critical for many heathland birds that are characterised by low reproductive rates and poor dispersal capabilities (Woinarski 1999a, 1999b). Furthermore, by maintaining a greater proportion of the landscape in mid- to late-

successional phases, more options would remain available to fire managers (e.g. planned burning) to implement strategies based on the current state of the system (Richards *et al.* 1999).

A large body of research to date, including results from this study (see Chapters 4-6), not only indicates that the extent of fires should be limited in relation to total suitable habitat area, but that burnt areas should also contain an adequate number, size, and interspersed of unburnt patches to provide refugia during and after fire events; facilitate persistence, dispersal, and post-fire recolonisation; and adequate reproduction over space and time (McFarland 1998b; Bryant 1991; Reilly 1991b; Little and Cutten 1996; Woinarski 1999b; Bradstock *et al.* 1995; Baker 2002; Garnett and Crowley 2002; Maguire 2005; Paton *et al.* 2005; Tasker and Baker 2005; Burbidge *et al.* 2007; Gibson *et al.* 2007; Bain *et al.* 2008; Lindenmayer *et al.* 2009). Such patches of unburnt or partially burnt habitat thus function as 'biological legacies' that facilitate persistence of pre-disturbance organisms, patterns, and processes, and thus the recovery of the avifauna in the post-disturbance environment, particularly in the case of large, intense fires (Franklin *et al.* 2000). The *Draft Ecological-management Burning Prescriptions* (PWS 2004) also contain a number of guidelines that, if implemented, should facilitate meeting some of these objectives; these should also be incorporated into hazard-reduction burning when feasible and consistent with other management objectives. These include burning < 90% of small sites (i.e. ≤ 100 ha) and < 50% of large sites (i.e. > 100 ha), maintaining unburnt patches (> 0.25 ha) no more than 500 m apart, and using a dispersed ignition pattern. However, results from this study suggested that burning up to these thresholds may not ensure persistence of the resident species, particularly at small sites and in the short-term (see Chapters 4 and 6).

The results from this study suggested that preservation of moorland riparian habitats is particularly important since they provide greater relative cover and food resources, and may serve as important post-fire refugia and dispersal corridors, particularly in the case of extensive fires that may displace large portions of the resident populations (see Chapters 4-6). The effects of prescribed burning operations on riparian zones have not been explicitly addressed to date (Marsden-Smedley *et al.* 1999; Marsden-Smedley and Kirkpatrick 2000; PWS 2004; Marsden-Smedley 2009). Parts of riparian areas may remain unburnt under some conditions; however, riparian areas can be extensively burnt by a primary fire (see Chapter 6), or may be hand-torched by fire crews if such areas of unburnt vegetation remain after the initial ignition and fire front have passed (T. Norris pers. comm. 2003; pers. obs.). Since burning such small patches of habitat is unlikely to significantly contribute to hazard-reduction aims, it is recommended that fire crews are informed of the importance of ensuring that portions of riparian habitats and other emergent vegetation (e.g. scrub copses) remain

unburnt within each site to provide post-fire refugia for the fauna. These recommendations should be considered in relation to back-burning operations when responding to wildfires as well (Lindenmayer *et al.* 2009).

Inter-fire intervals and frequency

Although data for all known fire events in the study area were presented in Chapter 2, fire frequencies were not calculated due to a lack of adequate data and the inability to verify the mapping of historical burns at the site scale. However, it is apparent that most sites have burnt at frequencies that are consistent with the maintenance of moorland communities, at least for blanket moorlands in southwestern Tasmania (i.e. 20-40 year inter-fire intervals; Jackson 1968, 1999a). Furthermore, part of the site selection process for this study included control of variation in fire frequencies and inter-fire intervals (i.e. > 10 years), at least over the past two fire events, to help minimise these variables from confounding the effects of time since fire *per se* (see Chapter 2). Accordingly, there is some uncertainty as to the potential effects of multiple, shorter inter-fire intervals of < 10 years on the resident species.

Current hazard-reduction prescriptions for moorlands suggest that the optimum interval between burns is 7-10 years (range 5-15 years) for low productivity sites (e.g. Lake Pedder) and 5-8 years (range 5-10 years) for medium productivity sites (e.g. Lake St Clair) (Marsden-Smedley *et al.* 1999; Marsden-Smedley 2009). Recommendations for ecological-management burning encompass a much broader range of inter-fire intervals, including short intervals of 20-30 years and long intervals $\geq 30^+$ years for low productivity sites, and short intervals of 5-12 years and long intervals > 12 years for medium productivity sites (Marsden-Smedley and Kirkpatrick 2000; PWS 2004; Pyrke and Marsden-Smedley 2005; Marsden-Smedley 2009). They also recommend that fire frequencies are varied, with a maximum of two short or two long consecutive burn intervals. Since the highest mean densities of the residents in low productivity moorlands were at the oldest sites surveyed (i.e. 54 years post-fire), the guidelines, particularly for hazard-reduction burns, would likely not be sufficient for resident populations to attain peak densities at some sites (see Chapter 4). Therefore, it is recommended that the majority of burns should follow the ecological-management guidelines, provided they include multiple burns of long intervals > 50 years. In contrast, the highest mean densities of the residents in moderate productivity moorlands were in younger sites (i.e. 2-8 years post-fire) (see Chapter 4). Therefore, guidelines for ecological-management burns appear to be appropriate to maintain resident populations in moderate productivity sites, provided that burns with short inter-fire intervals (i.e. < 10 years) are limited in extent and include adequate areas of unburnt vegetation.

For conservation of the Ground Parrot, Bryant (1991) recommended not burning more than 25% of an area over a 10 year rotation in order to maintain suitable habitat and sustainable densities across a landscape. This appears to be appropriate for eastern moorlands, but the frequency may be too high for blanket moorlands in which habitat > 50 years post-fire is likely needed to provide suitable conditions for populations to attain maximum densities within portions of the region (see Chapter 4). She also cautioned against ‘overfiring’ of moorlands in the Lake St Clair area since it represents an important eastern fringe of Ground Parrot distribution in Tasmania (Bryant 1991). While repeated hazard-reduction burning in some areas may cause adverse, although probably localised, impacts on the resident species (Bryant 1991, 1992; Little and Cutten 1996), this needs to be weighed against the reduced risk of intense and extensive wildfires across the landscape, which may ultimately pose the greatest threat to the avifauna (Brooker and Rowley 1991; Brooker 1998; Reilly 1991a; Loyn 1997; Marsden-Smedley and Kirkpatrick 2000; Burbidge 2003; PWS 2004; Burbidge *et al.* 2005; Pickett 2005; Marsden-Smedley 2009). King *et al.* (2006, 2008) recently modelled the effects of different prescribed burning strategies in moorlands compared to the risk of unplanned wildfires and assessed the potential impacts on ecological values in southwestern Tasmania. Simulations indicated that prescribed burning treatment levels of between 5% and 10% of total moorland area on an annual basis and consisting of small treatment units (i.e. a ‘fine-scale fuel mosaic’) would reduce the mean extent and incidence of unplanned fires and thus help to conserve fire-sensitive species and overall biodiversity. Treatment levels at the lower end of this range would represent a fairly modest increase from the current levels. However, the upper end would result in higher fire frequencies than were believed to characterise Aboriginal and early European fire regimes, and may adversely impact moorland ecosystems (Marsden-Smedley 1998b; Marsden-Smedley and Kirkpatrick 2000; PWS 2004; Pyke and Marsden-Smedley 2005; Driessen 2006; King *et al.* 2006, 2008).

Relevant recommendations from the mainland concur that inter-fire intervals of at least approximately 10 years and fire regimes of variable frequencies should benefit most bird species, but that longer intervals (e.g. > 20 years) may be required for some cover-dependent threatened species (Meredith *et al.* 1984; McFarland 1988b; Brooker and Rowley 1991; Garnett 1992; Little and Cutten 1996; Baker 1997; Woinarski and Recher 1997; Brooker 1998; Woinarski 1999a, 1999b; Garnett and Crowley 2002; Keith *et al.* 2002b; Burbidge 2003; Recher 2005; Tasker and Baker 2005; Burbidge *et al.* 2007). Furthermore, long intervals (i.e. 20-30 years) may be necessary to conserve peat soils, particularly if trends in global warming continue (Bridle *et al.* 2003). Although species associated with early successional habitats in some regions may be disadvantaged by such a fire regime (e.g.

Richard's Pipit and Flame Robin), these species are typically opportunistic, widespread, and only provide transient increases to local avian diversity (Woinarski and Recher 1997).

Monitoring

Proposed ecological-management guidelines contain a number of general recommendations for monitoring during and after fires that, if implemented, should provide critical data for adaptive fire management (e.g. burn weather conditions, area, patchiness, fuel removal) (PWS 2004; Marsden-Smedley 2009). It is recommended that such data are collected during all planned burning and wildfire suppression operations, whenever feasible, particularly considering the high variability in site-specific avifaunal responses to fire (Burbidge 2003; see Future directions). More specifically, fire crews should be provided with basic training on how to identify the resident species by sight and sound, and should record data on any opportunistic observations of avifaunal behaviour during and after any fires, as well as any evidence of post-fire mortality within burn areas. Such observations will be particularly important in assessing the potential for the resident species to escape the varied intensities and fire patterns resulting from different ignition methods and patterns (e.g. hand vs. aerial and line vs. point ignitions) (Parr and Chown 2003).

It is recognised that the intensive nature of the avian survey methods used for this study would not be feasible to conduct on a regular basis and across large areas. Therefore, from a practical standpoint, the most critical component of monitoring is to conduct annual presence-absence surveys of the resident species. Results from this study showed that the resident species appeared to be present in healthy numbers across the study area (see Chapter 4). However, the Ground Parrot was more patchily distributed in the study area, and the majority of moorlands in the TWWHA have not been surveyed to date. Ideally, surveys should be conducted throughout the TWWHA, or at the very least in areas that are the focus of fire management activities, such as within the Lyell Highway corridor near Lake St Clair (PWS 1996; see Chapter 2). Minimal training would be required to instruct staff on how to identify the resident species by sight and sound. During the course of this study it took only about 20 minutes on average to establish presence of the Southern Emu-wren and Striated Fieldwren during point and line transect surveys. Accordingly, only short and informal point and walking surveys would be required to establish occupancy by the resident species. Ground Parrot surveys would need to be conducted during either their dawn or dusk calling-flight sessions (Bryant 1991; see Chapter 4). However, the Southern Emu-wren and Striated Fieldwren are also often readily detectable at dawn and dusk (T. Chaudhry unpublished data). Such surveys would require minimal resources and could easily be conducted during the course of other management activities in moorlands (e.g. during pre- and post-burn assessments; Marsden-Smedley 2009). Alternatively, the use of automated sound recording

systems could also be considered (e.g. Cunningham *et al.* 2004). Although the information provided by simple presence-absence surveys is somewhat limited compared to intensive surveys such as distance sampling (see Chapters 4 and 6), considerable work has been recently conducted to maximise the inferences that can be made from such datasets for the purposes of wildlife management (e.g. Royle and Nichols 2003; Westphal *et al.* 2003; Brotons *et al.* 2004; MacKenzie 2005). However, in some cases it may be necessary to conduct repeated surveys over relatively short timeframes (i.e. a few within a season) in order to establish 'absence' at a given site with certainty and thus obtain more reliable parameter estimates (MacKenzie 2005). If the presence-absence surveys become standard practice, these data could be used to determine if fire management activities are leading to adverse effects, such as local extinctions, and then be used to adapt strategies accordingly (e.g. reduce frequency and extent, increase fine-scale patchiness). Ideally, these data would be used to build and validate models that would assist managers in predicting the potential impacts of different management strategies on the avifauna (see Future directions).

Adaptive management

The above recommendations for the conservation of the avifauna of buttongrass moorlands should be evaluated within an adaptive management framework, using a more flexible approach towards fire management that will allow strategies to be adapted as both conditions and understanding of fire ecology change (Bradstock *et al.* 1995; Gill *et al.* 2002; Keith *et al.* 2002b; Kotliar *et al.* 2005; Tasker and Baker 2005). This will necessitate the use of a more systematic and ecosystem-oriented approach, which will need to include clear management objectives based on the current state of the landscape and desired future conditions, sound science, risk assessment, long-term monitoring, and ongoing formal evaluation (Richards *et al.* 1999; Gill *et al.* 2002; Keith *et al.* 2002b; Bradstock *et al.* 2005; Collins 2006). This entire process must be flexible enough to adapt to unexpected outcomes (e.g. on non-target species and food resources), unplanned wildfires, and other possible disturbances (e.g. climate change) (Bradstock *et al.* 1995; Keith *et al.* 2002b; Lindsay 2003). As far as possible, indigenous knowledge of fire should also be incorporated into management strategies. However, it should be recognised that such strategies were most likely spatially and temporally variable, and that significant gaps in knowledge of the historical range and variability of fire regimes will likely persist. Accordingly, information on historical range and variability should only be used as a guide, and not to try and recreate an uncertain past in the highly-altered and changing modern landscape (Marsden-Smedley and Kirkpatrick 2000; Keith *et al.* 2002b; Burbidge 2003; Kotliar *et al.* 2005; Collins 2006).

The general strategy outlined above is based to some extent on the 'mosaic paradigm' and is essentially a form of bet-hedging; the hope is that providing a range of habitat conditions

across the landscape will result in the availability of adequate resources for most species over time (Bradstock *et al.* 1995). However, at its core is the ability to adapt to and incorporate natural variability and to manage fire in a way that does not attempt to impose an ideal state across the landscape. Rather, it allows ecosystems to operate within critical thresholds based on the best available science on both the historical and the future range of variability and management objectives (Bradstock *et al.* 1995; Gill *et al.* 2002; Bradstock *et al.* 2005; Keane *et al.* 2009). The ability to adopt an adaptive management approach within Tasmanian buttongrass moorlands is predicated on a continued commitment to conservation and adequate funding, research, and monitoring (see Future directions).

Future directions

This study has identified some key areas where more research is required in order to develop a better understanding of avifaunal dynamics in the pyrogenic buttongrass moorlands of Tasmania. In many respects, future research in buttongrass moorlands should focus on the fundamentals of avian population biology and community ecology, since many of the questions raised in this study can only be adequately addressed by developing a better understanding of the life cycle processes of the avifauna, particularly of the resident Southern Emu-wren, Striated Fieldwren, and Ground Parrot. However, due to the difficulties associated with studying such extremely cryptic species, particularly within a landscape context, long-term political commitment and considerable resources will be required to obtain enough quality data on which to base prudent management decisions. Such a proactive approach is certainly warranted within a World Heritage ecosystem (PWS 1999); although, in reality such efforts are often reactive and limited to species that are already threatened. Until such studies are undertaken, knowledge of the Tasmanian buttongrass moorland avifauna, and particularly of the resident species, will remain as cryptic as the birds themselves. Despite the obvious challenges that lay ahead of us, some of the major gaps in our current knowledge and some specific suggestions for future research directions are provided below.

Construction and verification of fire regimes

Since reliable and consistent fire history data are still lacking for Tasmanian buttongrass moorlands, similar to many other regions of the world, future studies exploring moorland fire ecology will depend on continued efforts to verify existing fire history records and to characterise and map future fire events (Parr and Chown 2003; Bridle *et al.* 2003; Bradstock *et al.* 2005; Tasker and Baker 2005; PWS 2004 unpublished data). Since most of the GIS mapping of historical fires was necessarily conducted at a coarse scale (Marsden-Smedley 1998a; PWS unpublished data), it is important to ground-truth the data within moorlands using the methods described in Chapter 2, as well as to assess the efficacy of alternative

methods for constructing long-term fire histories for the region (e.g. dendrochronological studies in adjacent forests). In addition, although a greater emphasis has been placed on fire record keeping in recent years, detailed data on ignition methods and patterns, fuel loads, fire behaviour, and burn patchiness should be collected for all prescribed burning operations conducted by Parks and Wildlife Service and Forestry Tasmania, as well as for wildfires when feasible (PWS 2004; Marsden-Smedley 2009). Particularly if spatially-explicit (e.g. based on remote sensing) and provided in a readily accessible format (e.g. online GIS database), such data will not only be instrumental for continued fire ecology research, but would also help to validate and refine the fuel and fire behaviour models in current use by fire managers (i.e. Marsden-Smedley *et al.* 1999; Marsden-Smedley 2009). In the long-term, detailed and accurate fire history records will be invaluable in investigating the potential impact of different fire regimes on moorland ecosystems, particularly in relation to fire behaviour, scale, and frequency.

Robust experimental design

During the planning and implementation stages of this study, considerable effort was expended to ensure that the design was appropriate for addressing the primary research aims, included an adequate range of fire ages and replication (Baker and Whelan 1994; Whelan 1995), and investigated the influence of key abiotic and biotic variables in addition to time since fire *per se* (Woinarski 1999b; Smith 2000; see Chapter 2). However, this study was limited by similar time and resource constraints as imposed on most doctoral research projects. Although most fire ecology studies to date have not included any replication (Whelan 1995; Whelan *et al.* 2002), the inclusion of more replicates in this study may have increased power and provided additional insights into the dynamics of moorland ecosystems. In order to develop a more process-oriented understanding of pyrogenic ecosystems, it will be necessary to conduct long-term, manipulative experiments (Whelan *et al.* 2002; Ward and Paton 2004a; Saab and Powell 2005). This will not only require ongoing commitment and funding (e.g. Brooker 1998), but also close and flexible collaboration between researchers and land management authorities, especially considering the often narrow windows during which prescribed burns can be conducted (Whelan 1995; Marsden-Smedley *et al.* 1999; Marsden-Smedley 2009).

Demographics

To date no formal demographic studies have been conducted on the three moorland resident species in Tasmania and only limited research has been conducted on mainland populations. Notable examples are Pickett's (2000) and Maguire's (2005) research on the Southern Emu-wren in South Australia and Victoria, respectively. However, no similarly detailed demographic studies have been conducted on the Ground Parrot or Striated Fieldwren (for

reviews see Higgins 1999; Higgins and Peter 2002). This dearth of research is not surprising given the resource-intensive nature of demographic studies, particularly for cryptic, ground-dwelling species that are rarely seen, making colour-banding of limited utility. The task becomes even more daunting when trying to examine demographic trends within a fire and landscape ecology framework (e.g. Brooker 1998). Nevertheless, even obtaining baseline demographic data on Tasmanian populations would help to provide a fitness framework within which to consider the findings of this and subsequent studies, and would contribute to a better understanding of the underlying processes (Morse 1990; Wolda 1990; Alldredge *et al.* 1998; Jones 2001; Whelan *et al.* 2002; MacHunter *et al.* 2009). Ideally, such information, along with that obtained during this study, will enable the development of spatially-explicit, fitness-based models that will make it possible to predict species densities and diversities across the landscape, and ultimately the potential risks of different management regimes on the avifauna (e.g. Brooker and Brooker 1994; Littley and Cutten 1996; Richards *et al.* 1999; Westphal *et al.* 2003; Buchanan *et al.* 2005; Bradstock *et al.* 2005; MacHunter *et al.* 2009). An adequate amount of quality autecological data is needed to parameterise accurate models since predicted outcomes are likely to be highly sensitive to even small discrepancies between expected and observed attributes of species (Keith *et al.* 2002a; Whelan *et al.* 2002).

Accordingly, some areas of research of particular value include the identification of specific breeding periods of the resident species in relation to climate change and investigating fire-related mortality, site fidelity, territoriality, dispersal, recolonisation, reproduction, and population levels (Whelan 1995; Gill *et al.* 2002; Keith *et al.* 2002b; Whelan *et al.* 2002; Bradstock *et al.* 2005; Maguire 2005; Paton *et al.* 2005; MacHunter *et al.* 2009). Such studies would undoubtedly be facilitated by the use of radio-telemetry, which has already been successfully used on the Ground Parrot for at least short periods of time (Jordan 1987b; McFarland 1991a). However, transmitters are yet to be developed that are small enough to be used on diminutive birds such as the Southern Emu-wren and Striated Fieldwren (Maguire 2006a).

Foraging behaviour and dietary preferences

Results reported in Chapter 5 indicated that potential arthropod prey resources (i.e. order-level abundance and energy content) for insectivorous species varied in relation to both habitat type and fire age in eastern moorlands at Lake St Clair, while related research conducted on moorland invertebrates at Lake Pedder indicated that similar patterns may exist in blanket moorlands (e.g. Greenslade and Driessen 1999; M. Driessen unpublished data; see Chapter 5). However, information used in this study regarding the foraging behaviour and dietary preferences of the insectivorous species was largely drawn from the literature and

based on studies conducted in other regions of Australia. Hence, it would be worthwhile to conduct research in Tasmanian moorlands, across a chronosequence of sites if possible, focused on identifying the foraging behaviour and dietary preferences of the Southern Emu-wren and Striated Fieldwren. Given their cryptic nature, detailed behavioural observations are extremely difficult to obtain (see Chapter 5), so they will probably need to be supplemented with other methods such as remote viewing of food provisioning at nest sites (e.g. Maguire 2006b), stomach content analyses (e.g. Lea and Gray 1936), and stable isotope analyses (Hood-Novotny and Knols 2007). A component of this research should also include obtaining additional food resource data (e.g. at Lake Pedder) of a finer taxonomic resolution and encompassing multiple seasons. Additional avenues of research include the identification of other factors that may influence habitat choice, such as site fidelity and territoriality, sexual differences, competition, and predation (Cody 1981, 1985; Jones 2001).

Interspecific interactions and niche differentiation

Based on the results from this study it is clear that the insectivorous and ground-dwelling Southern Emu-wren and Striated Fieldwren share very similar niches and are sympatric throughout much of their ranges in western Tasmania, with the latter occurring in a somewhat broader range of sites (see Chapters 4-6). Although such associations are not uncommon, their's is of particular interest since they are the only two resident insectivores found in over 0.5 million hectares of Tasmanian moorlands. How are these species able to coexist in a habitat characterised by limited structural diversity and overall productivity? Although these species have been seen in close proximity to each other (i.e. < 2 m) while showing no obvious agonistic behaviour (pers. obs.), essentially nothing is known about the nature and extent of interspecific interactions between these species. Are niches differentiated through the maintenance of spatially exclusive territories, or through fine-scale differences in habitat selection and resource use? In particular, what role do interspecific interactions and niche differentiation play in the competitive post-fire environment where cover, food, and nesting resources are often in short supply? As with the other suggested avenues of research, attempting to answer these questions will be extremely difficult. Some possible approaches may include intensive behavioural observations, territory mapping, and radio-telemetry, if and when feasible.

Potential threats

Although populations of the Southern Emu-wren, Striated Fieldwren, and Ground Parrot are not currently listed as threatened in Tasmania, some of the lessons learned from research on mainland populations indicate that potential threats, in addition to altered fire regimes, do exist in Tasmania. The most likely direct threat comes from native predators (e.g. Tiger Snakes, *Notechis scutatus*), and particularly from feral predators such as cats (*Felis catus*)

and Red Foxes (*Vulpes vulpes*), and from possible interactions between predation and fire disturbance (Burbidge 2003). Feral cats have been sighted in both the Lake Pedder and Lake St Clair areas and are known to prey on resident species in Tasmania (Bryant 1991; PWS unpublished data; pers. obs.), as well as populations on the mainland (Pringle 1982a; McFarland 1991c; Maguire and Mulder 2004; Maguire 2005). Recently, feral foxes were illegally introduced into Tasmania and they may pose a significant threat to the moorland avifauna (Saunders *et al.* 2006), particularly if they are able to expand into the TWWHA. Although it would be interesting to investigate current and potential future impacts of feral predators on the avifauna, resources would be better used to expand current eradication efforts since the impacts of feral predators on the Australian fauna are well documented (Dickman 1996).

Other potential threats to the moorland avifauna that should be examined include the impact of the introduced water mould and plant pathogen, *Phytophthora cinnamomi*, on avian habitat and food resources (Bryant 1991; Brown *et al.* 2002; Schahinger *et al.* 2003). These effects may be confounded with or exacerbated by those of different fire regimes (Podger 1990; Brown *et al.* 2002), and should be explicitly considered when planning and implementing fire management activities (PWS 2004). Finally, climate change predictions indicate the possibility of adverse impacts on the Australian fauna, which may be exacerbated by altered fire regimes (e.g. increased frequency and severity) (Watson *et al.* 1999; Cary 2002; Bridle *et al.* 2003; Lindesay 2003). The only way to assess the possible effects of climate change on the Tasmanian moorland avifauna and to adapt fire and conservation management strategies accordingly, is to conduct long-term monitoring of avian populations and habitat (Bryant 1991; Whelan 1995; PWS 1999; Baker 2000; Tasker and Baker 2005).

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