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Movement, habitat requirements, nesting and foraging site selection: a case study of an endangered granivorous bird, the Black-throated finch *Poephila cincta cincta* in north-eastern Australia

Thesis submitted by

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A Black-throated finch southern subspecies (*Poephila cincta cincta*) fitted with a radio-tracking device on the Townsville Coastal Plain, Queensland, Australia.

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Thesis Abstract

Species all over the world are declining to unsustainable population levels due to habitat destruction, introduced species and pollution. Susceptible species begin to decline prior to noticeable degradation of the environments in which they live. Savannas are the dominant vegetation type in northern Australia and, although considered relatively unmodified, have experienced severe faunal decline, particularly among granivores. Granivorous birds represent 20% of Australian land birds and range-scale declines have been documented for several species. Declines of the granivorous birds of Australian tropical savannas are associated with land use change resulting from European settlement, such as land clearance, changes in fire regime, pastoralism, introduced species and shrub proliferation.

Although it is generally recognized that granivorous birds have declined, there are still many gaps in our understanding of the underlying causes. The Black-throated finch southern subspecies (*Poephila cincta cincta*, herein referred to as BTF) is an endangered, endemic granivore of eastern Australia. BTF have had an estimated range contraction of 80% since the late 1970s. Little is known about BTF ecological requirements, home range size or movement patterns. Increased knowledge of these will greatly assist with management and conservation actions. In this thesis, I investigate the ecology of BTF in north-eastern Australia. I examine (a) home range sizes, movement patterns and habitat use, and (b) habitat requirements, nesting and foraging site selection.

Understanding how animals move in the landscape to meet their demands – food, water, and shelter - is a prerequisite for successful conservation outcomes. To acquire more information about BTF movements, I mist netted in eight sites on the Townsville Coastal Plain (TCP) and radio-tracked in two of these sites. I color-banded 102 BTF and estimated the home ranges of 15. More than half of all resightings occurred within 200m of the banding

site and within 100 days of capture. Long distance movements (up to 17 km) were recorded for only three individuals. Home range size differed between sites but not between seasons (early dry season and late dry season). BTF home ranges encompassed four broad vegetation types among eight available (Regional Ecosystems), with habitat selection significantly different from random. BTF showed a distinct local daily movement pattern at one site (roosting to feeding area). In this study, BTF maintained home ranges ranging from 25.2 to 120.9 ha over short time scales (e.g. within seasons).

Vegetation structure and composition greatly influence the way animals use habitat. Determining key habitat features for a threatened species is paramount for identifying appropriate management actions. To understand these for BTF, vegetation surveys were undertaken at 10 sites on TCP. BTF flocks were most closely associated with higher cover of native grasses, low shrub cover, with the presence of dead trees and high cover of certain grass species, e.g. *Eragrostis* spp. Small flocks were associated with low percentages of native grasses, high shrub cover and low grass richness. BTF showed a preference for nesting at sites with lower tree and grass diversity, but no preferences for ground cover structural features. Other environmental structural features in the nesting habitat might be more important to the birds.

I explored BTF nesting habitat selection by comparing areas around nests (used) with that in the surrounding area (available). Individual nests were used for breeding and/or roosting. Fifty active BTF nests were found during this study. BTF nested in four tree species, preferentially using *Eucalyptus platyphylla* and *Melaleuca viridiflora* in areas of low tree density. BTF showed a preference for nesting in sites with lower tree and grass diversity, but no preference for ground cover structural parameters. Other environmental features in the nesting habitat might be more important to the birds as structural features.

Specific patches where animals are exploiting resources differ in nature and appearance from the matrix in which they are embedded. Being ground-foragers, BTF are likely to be specific in the micro-structure preferences of foraging patches, such as the presence of bare ground patches. To understand the BTF needs, foraging patches – specific areas where animals were exploiting resources – were examined. Vegetation structure and composition were compared between foraging patches (used) and surrounding (neighboring) and general areas (available). Of the ground cover structural characteristics, nearly all variables were significantly different between used, neighbouring and available areas. BTF foraged preferentially in areas with lower diversity of grasses but nearby areas with high diversity (neighbouring areas). BTF selected specific structural vegetation features for foraging patches, compared with neighboring and available areas, particularly the ground cover features. Foraging patches were less densely vegetated than neighboring and available areas; however, they adjoined areas with high grass structural complexity (higher visual obstruction, higher vegetation density in almost all levels measured, higher vegetation cover, particularly grass cover, and higher number of species per hectare). In summary, foraging BTF require open areas finely interspersed with to grassy areas.

In this study, I found that BTF habitat must encompass patches with suitable grasses (e.g. *Eragrostis* spp.), and patches with bare ground or low vegetation density (ground cover) to allow BTF access to the seed bank. BTF prefer a general absence of shrubs but the scattered presence of a medium strata. Therefore, large homogeneous areas will not usually meet the requirements of BTF populations. Extensive woody thickening could disadvantage BTF, as has been found for other granivorous birds such as the Golden-shouldered parrot (*Psephotus chrysopterygius*). Instead, they require a mosaic of vegetation within their daily home range: areas with bare ground finely interspersed with areas of suitable grass species, low shrub density, presence of suitable woody plant cover and the presence of species such as *Eucalyptus platyphylla* and *Melaleuca* spp.

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Chapter 1

General Introduction

Worldwide species declines

The tropical grassy biomes

Worldwide, many species are declining to unsustainable population levels as their habitat is destroyed, fragmented and impacted by pollution, introduced species and human activities (Hilton-Taylor 2000, Zedler et al. 2001). The monitoring of status and trends of biodiversity is still deficient, leading to a lack of knowledge of species' vulnerabilities (Hilton-Taylor 2000). Sensitive species may begin to decline prior to any noticeable degradation of the habitat in which they occur (Shaffer 1981). Species susceptibility to extinction can be influenced by population size, range size, adaptability to new conditions, or reproductive output (Reside et al. 2015). Loss of species affects ecosystem functioning and persistence, including the provision of ecosystem services essential to human well-being (Zedler et al. 2001, Balvanera et al. 2006). Although many studies have investigated species loss in tropical forests (Turner 1996, Corlett 2014, Ochoa-Quintero et al. 2015), tropical grassy biomes have attracted less conservation attention (Parr et al. 2014).

Tropical grassy biomes - savannas and grasslands - dominate the tropics (Bourlière and Hadley 1983, Solbrig et al. 1996, Bond and Parr 2010), covering about 20% of the world's land surface (Cole 1986, Scholes and Archer 1997, Franklin 1999, Sankaran et al. 2005, Parr et al. 2014). They are globally important to human economies, supporting a large proportion of the world's human population and most of its rangeland, livestock and biomass of large, wild herbivore (Scholes and Archer 1997, Sankaran et al. 2005, Bond and Parr 2010, Parr et al. 2014), providing important ecosystem services and influencing the earth's-atmosphere (Bond 2008, Parr et al. 2014). Savannas are typical in areas where rainfall is highly seasonal (Johnson and Tothill 1985, Solbrig et al. 1996, Williams et al. 1999).

However, savannas and grasslands have undergone a great reduction in extent due to human activities and are facing substantial conservation threats (Werner 1991, Fearnside 2001, Thomas and Palmer 2007, Eriksen and Watson 2009, Bond and Parr 2010, Parr et al. 2014).

Grassy biomes and decline of granivorous birds in Australia

Savannas are the dominant vegetation type of northern Australia (Wilson et al. 1990, Gillison 1994), extending from south-eastern Queensland across the north of the continent (Solbrig et al. 1996), and covering almost a quarter of Australia (Williams and Cook 2001b). Australian savannas range from open forest to woodland, open woodland and grassland (Williams and Cook 2001b). Savanna woodlands, for example, occur in the north of Western Australia and the Northern Territory, in eastern Queensland, extending southwards to New South Wales (Cole 1986). The variation in vegetation types across Australian savannas are driven by rainfall and soil patterns (Williams et al. 1996, Williams and Cook 2001b) and, although eucalypts (*Eucalyptus* spp., *Corymbia* spp.) and *Acacia* spp. are the main trees throughout, these two factors determine the particular tree species that occur and the associated grasses (Williams and Cook 2001b).

Although the general appearance and vegetation structure of Australian savannas are considered relatively unmodified (<1% cleared and <0.01 people per km^2), their fauna has experienced substantial declines (Franklin et al. 2005, Woinarski et al. 2011, Edwards et al. 2015). Many species of medium-sized mammals and granivorous birds have declined or become regionally extinct during the past few decades (Woinarski and Catterall 2004, Woinarski et al. 2004, Woinarski et al. 2007, Woinarski et al. 2010, Woinarski et al. 2011, Woinarski and Legge 2013, Edwards et al. 2015). Granivorous birds comprise 20% of all Australian land birds. They are prominent in lists of threatened Australian threatened bird taxa (Franklin et al. 2000) suggesting broad-scale historical declines (Garnett and Crowley

2000c, Franklin et al. 2005, Garnett et al. 2011a). Examples of declining granivores include the extinct Paradise parrot (*Psephotus pulcherrimus*), the endangered Golden-shouldered parrot (*Psephotus chrysopterygius*), the endangered Gouldian finch (*Erythrura gouldiae*) and the Black-throated finch (*Poephila cincta*) with the subspecies *P. c. cinta* listed as endangered federally and at a state level (EPBC 1999, Garnett and Crowley 2000c).

The declines of the granivorous bird assemblage of Australian tropical savannas are associated with European settlement in the last century (Franklin 1999, Franklin et al. 2005, Woinarski and Legge 2013). Vegetation clearing, introduced species, changes in fire regimes and land use are the main factors believed to be responsible (Franklin et al. 2005, Edwards et al. 2015); they can act individually or in concert. Land use is now a patchwork of pastoral, conservation, indigenous, military and mining activities (Williams and Cook 2001b). The primary processes affecting savannas (fire regime, grazing, land clearing, urbanization, species introductions) result in changes in the density and nature of watering points, changes in the tree-grass balance, and general changes in vegetation composition. These factors affect the landscape in a very complex interplay of processes (Fig. 1.1).

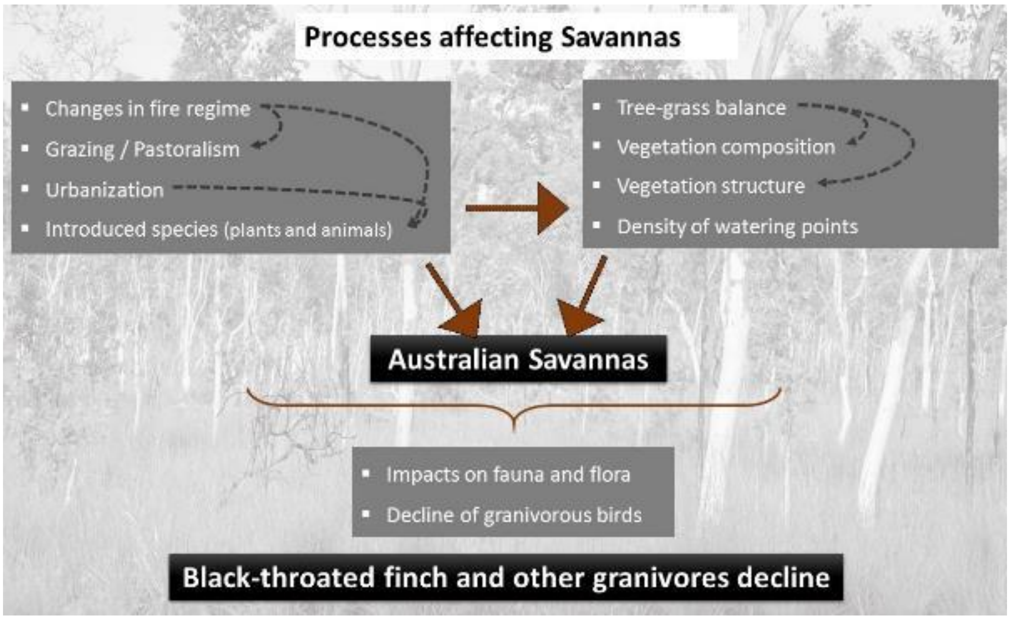

Fig. 1.1. Processes affecting savannas landscapes and interactions among them and with granivorous birds in Australia.

Land use changes in Australian savannas – Land clearing

Land clearing is the greatest threat to terrestrial biodiversity worldwide (Hannah et al. 2007) and in Australia (Saunders et al. 1991b, Rolfe 2002). Land clearing impacts ecosystems by killing biota and removing habitat, fragmenting populations, destabilizing ecological process and reducing ecosystem resilience (Gibbons and Lindenmayer 2007). Most of the land clearing in Queensland has occurred in the last 50 years, largely for the cattle industry (Bradshaw 2012). Queensland was considered a land clearing hotspot between 1981 and 2000, with more than 80% of all land clearing in Australia (Bradshaw 2012). In Queensland, two-thirds of the natural vegetation has been cleared in the past 150 years, initially for agriculture and pastoralism but increasingly for residential development (Sewell and Catterall 1998). Most of the land clearing in Queensland happened in the Brigalow Bioregion, with most taking place in the 1960s (Bradshaw 2012). More than 50% of land clearing in tropical Queensland since European colonization was associated with production of sugar cane, bananas and livestock; more than 50% of the northeast wet tropics region is under pasture (Bradshaw 2012). In contrast, the Northern Territory has experienced the least amount of land clearing; though the region has experienced altered fire regimes, proliferation of non-native vegetation and pastoral activities (Bradshaw 2012). Clearing of native vegetation has a notable influence on bird assemblages, with cleared habitats containing different and depauperate bird assemblages (Martin and McIntyre 2007). An estimated 8.5 million of birds die each year in Queensland due to land clearing, many being woodland birds, including parrots and finches (Cogger et al. 2003). Many species do not survive in disturbed habitats (Cogger et al. 2003), and this particularly true for granivorous birds (Franklin 1999). Even where land has not been extensively cleared, more subtle forms of modification are threatening savanna biota.

Land use changes in Australian savannas - Fire

Fire is a major determinant of biodiversity patterns in savannas landscapes (Davis et al. 2000, Brawn et al. 2001, Reside et al. 2012a, Taylor et al. 2012). Fire affects (1) plant phenology (influencing the timing of flowering), and therefore (2) fruit and seed production (both intensity and timing), (3) seed availability, (4) seedling regeneration (Setterfield 1997), (5) resource production and accessibility (Woinarski et al. 2005), (6) soil erosion, (7) relative abundance of plant species, leaving just adult individuals (senescent effect) or by favouring a particular subset of species leading to changes in local diversity (Tran and Wild 2000). These effects on vegetation depend on fire intensity and frequency; and changes will influence plant species composition and structure (Russell-Smith et al. 2001, Russell-Smith et al. 2003, Woinarski et al. 2005). Intensity of savanna fires depends on fuel load, fuel moisture and wind speed (Andersen et al. 2003). Fires in the early dry season tend to be low intensity, patchy and limited in extent, while in the late dry season fires tend to be high intensity, burn extensive areas, and are more difficult to control (Williams and Cook 2001a, Andersen et al. 2003). Fire frequency affects savanna functioning by influencing the tree grass balance (Scholes and Archer 1997, Davis et al. 2000, Driscoll et al. 2010). High fire frequency prevents shrub proliferation (Roques et al. 2001), while low fire frequency may increase the risk of high intensity fires because of high grass biomass (Williams et al. 1999). Australian savannas have been characterized by frequent fire during their evolutionary history (Andersen et al. 2003) and this has had a dominant influence on the Australian landscape.

Fires regimes have involved both natural and anthropogenic burning (Tran and Wild 2000). Aboriginal people used fire widely across most of Australia (Williams and Cook 2001b) and these traditional burning practices persisted for thousands of years (Preece 2002, Gott 2005). Aboriginal fire regimes involved controlled and patchy fires which shaped the fauna, flora and structure of ecosystems (Gott 2005). European settlement caused significant change to the fire patterns; with a shift toward frequent, extensive, high intensity fires late in the dry season in northern savannas (Burbidge and McKenzie 1989, Williams and Cook 2001a, Andersen et al. 2003). Extensive and intense late dry season fires result in a homogenization of the landscape (Woinarski et al. 2005). Large and uniform areas, burnt or unburnt, tend to reduce biodiversity and can lead to regional or range-wide extinctions (Williams and Cook 2001a). A fine-scale mosaic consisting of areas subject to different intensities and frequencies of fire is purported to be ideal for maintaining the landscape-scale species diversity of savannas (Woinarski et al. 1999, Russell-Smith et al. 2001). Altered fire regimes have contributed to the extinction of two of the three bird species and three of the four subspecies which have disappeared from Australia since European colonization (Woinarski and Recher 1997, Woinarski et al. 1999). In addition, 51 bird taxa are recognized as threatened due to inappropriate fire regimes (Woinarski et al. 1999). Fire affects granivore dynamics in many ways as seeds must be available and accessible to the birds: a dense layer of grass or a vast area burned without resources available will affect them negatively; while less intense, patchy fires might expose the seed bank and therefore have benefits for granivores. Therefore, maintaining suitable fire regimes are important in biodiversity conservation (Driscoll et al. 2010).

Land use changes in Australian savannas - Pastoralism

In northern savannas, pastoralism is the dominant land use and the most widespread and long-standing exogenous disturbance to grasslands (Tremont 1994, Woinarski and Ash 2002, Martin et al. 2005). Pastoralism can lead to major changes in ecological function (Gifford and Hawkins 1978, Pettit et al. 1995). The most evident disturbance involves changes to the composition and structure of the ground flora (Whalley et al. 1978, Leigh and Holgate 1979, Pettit et al. 1995, Fensham and Skull 1999, Woinarski and Ash 2002, Kutt and

Woinarski 2007). Some of the compositional changes include: (i) increasing exotic grasses, (ii) loss of the most palatable grasses through over-grazing, (iii) changes to the species that dominate the herbaceous layer or (iv) decrease in species richness (Whalley et al. 1978, Tremont 1994, Pettit et al. 1995, Clarke 2003). An example of structural changes is the conversion of woodlands into grasslands by preventing recruitment of trees and shrubs (Pettit et al. 1995, Clarke 2003). The effects of grazing on bird fauna are complex, being confounded with other disturbances such as tree clearing, the provision of watering sources (Martin and McIntyre 2007) and its interactions with fire (Kutt and Woinarski 2007). Seed eaters are particularly affected because grazing by livestock often reduces the abundance of grass and seeds (Ford et al. 2001). Additionally, grazing alters grass species composition, removing key plant species and severely degrading granivore habitat (Lindenmayer and Burgman 2005).

Land use changes in Australian savannas - Urbanization

Despite the fact that Australian savannas cover 1.9 million square kilometres extending from Western Australia to Queensland, only 3% of Australia's population live there, most of the savannas being sparsely populated (Stoeckl and Stanley 2007). Additionally, Australian savannas are considered the largest intact area of natural vegetation in Australia (Franklin et al. 2005). Nonetheless, urbanization can be a threat to some species (e.g. the Black-throated finch southern subspecies *Poephila cincta cincta* (BTFRT/NRA 2011, Garnett et al. 2011a, Vanderduys et al. 2016)). It endangers species by replacing natural habitat with vegetation that does not provide essential resources (McKinney 2002), significantly changing the landscape (Gonzalez-Abraham et al. 2007). The natural vegetation is replaced by ruderal vegetation (cleared lots, farmland), or managed vegetation (residential, commercial) and a built environment (buildings and sealed surfaces) (McKinney 2002, Crooks et al. 2004, White et al. 2005). Trends in the physical environment along this gradient (human density, road density, air and soil pollution) reduce the habitat available for native species (McKinney 2002). Native species richness tends to decrease along this gradient (Beissinger and Osborne 1982, McKinney 2002, Crooks et al. 2004) while introduced species increase in abundance and richness (Jones 1981, Green 1984). Urbanization homogenizes bird communities (Sewell and Catterall 1998, Luck and Smallbone 2011, Aronson et al. 2014). Australia's human population is highly urbanized (O'Keeffe and Walton 2001, White et al. 2005), with over 85% of its citizens residing in urban centres and the transition to highly modified urban environments has occurred relatively rapidly (approximately 200 years) (White et al. 2005). Queensland holds the two remaining known populations of the endangered Black-throated finch southern subspecies (central Queensland and Townsville region, more details on the 'Black-throated finch' section on page 14). Although Queensland is a relatively decentralized State, with only 45% of the population living in the Brisbane region, 81% of the population still lives in areas defined by the Australian Bureau of Statistics as urban (O'Keeffe and Walton 2001). For instance, Townsville had a population growth of 2.5% between 2000 and 2005, compared with 2.2% for Queensland and it is projected to have the largest population growth in regional Queensland (Government Statistician Queensland Treasury and Trade 2013, Queensland Treasury and Queensland Government Statistician's Office 2015). In Australia, studies of urbanization and avifauna showed differences in species abundance and composition from suburban to native habitats (Wood, 1996, Jones 1981, Green 1984). In urban areas, introduced species (e.g Common myna *Acridotheres tristis*, Common starling *Sturnus vulgaris* and Feral pigeons *Columba livia*) often outcompete the native species for access to resources, such as food or tree cavities for roosting or nesting, and will displace them (Pell and Tidemann 1997, Sol et al. 2012, Grarock et al. 2013, Grarock et al. 2014). Additionally, aggressive native species such as the

Noisy miner (*Manorina melanocephala*), Australian raven (*Corvus coronoides*) and Australian magpie (*Cracticus tibicen*) exclude some species from their territories (Parsons et al. 2006, Sol et al. 2012). Some granivorous species, such as the Peaceful dove (*Geopelia striata*) and the introduced House sparrow (*Passer domesticus*) and Nutmeg manikin (*Lonchura punctulata*) are common in urban areas, while others, such as the Black-throated finch (*Poephila cincta*) seem to avoid even peri-urban areas (Jones 1983, Whatmough 2010). Urbanization is also known to facilitate non-native species, such as invasive grasses, that have important effects on an area's ecology (Vitousek et al. 1997, Fairfax and Fensham 2000).

Land use changes in Australian savannas – Introduced species

Introduced plants and animals can alter the environments they invade, modifying the composition and function of the native communities (Vitousek et al. 1997, Davis 2003, Mashhadi and Radosevich 2004) and reducing local diversity (Enserink 1999, Davis 2003, Grice 2006). Since European settlement in Australia, native animal and plant species have had to compete with a range of introduced species, contributing to the extinction of some and threatening some ecosystems (Benson 1991). Introduction of plants to Australia, along with vegetation clearing, livestock grazing and soil degradation, is one of the main threats to native species and communities (Grice 2004, Grice 2006, Grice et al. 2013). Replacement of native grasses by exotic species (e.g. Buffel grass *Cenchrus ciliaris*, Grader grass *Themeda quadrivalvis* and Guinea Grass Megathyrsus *maxima*) and crops has altered the composition and variety of seeds available to birds, favouring some species but being detrimental to others (Ford et al. 2001). The Red-rumped parrot (*Psephotus haematonotus*), for instance, feeds largely on seeds of exotic grasses and herbs, four of which are of particular importance because they provide a continuous food resource in winter (Lowry and Lill 2007).

Conversely, some granivores cannot survive on exotic species (Grice et al. 2013). Invasive grasses 1) are directly associated with pastoral development, 2) will dominate the understorey and thus reduce local grass diversity, 3) have seeds that might be less adequate for some granivorous birds and 4) result in more intense, extensive and frequent fire due to their high biomass. Bird species that may be negatively affected by invasive grasses via these various process are Masked finch (*Poephila personata*), Long-tailed finch (*Poephila acuticauda*), Hooded parrot (*Psephotus dissimilis*), Squatter pigeon (*Geophaps scripta*) and the threatened Black-throated finch (*Poephila cincta*), Gouldian finch (*Erythrura gouldiae*) and Golden-shouldered parrot (*Psephotus chrysopterygius*) (Grice et al. 2013). Introduced vertebrates prey on or compete with native fauna, graze native plants and alter community structure and food webs (Bomford 2003). Introduced predators, such as feral cats (*Felis catus*), are thought to have contributed to the extinction of 12 of Australia's 25 extinct bird taxa and negatively affect more than a third of Australia's 261 threatened bird taxa (Garnett and Crowley 2000c, Olsen et al. 2006, Garnett et al. 2011a).

Land use changes in Australian savannas – Woody thickening

Woody thickening refers to the increasing density of trees and shrubs and is usually associated with a reduction in grass production (Archer 1990, Moreira 2000). Woody thickening has been recorded in many areas of Australia since European settlement (Russell-Smith et al. 2001). In addition to changing the structure and composition of the vegetation, woody thickening can also increase the risk of soil erosion, disrupt nutrient cycling, and although it can be beneficial to some species, for example some insectivores (Tassicker et al. 2006, Seymour and Dean 2010), it can reduce habitat for other species less tolerant to dense vegetation, including many granivores (Gifford and Hawkins 1978, Pettit et al. 1995, Tassicker et al. 2006, Seymour and Dean 2010). Its occurrence is influenced by both fire and grazing regimes. Grazing can promote woody thickening by reducing grasses, spreading seeds of woody plants and reducing herbaceous fuel loads; hence lowering fire intensity (Russell-Smith et al. 2001). Less intense and less frequent fires do less damage to the woody vegetation, promoting recruitment and establishment of woody species (Knoop and Walker 1985, Brown and Archer 1988, Harrington 1991, Archer 1995, Roques et al. 2001, Van Langevelde et al. 2003, Beringer et al. 2007). Intense and frequent fires benefit grasses and suppress the recruitment of trees (Roques et al. 2001, Russell-Smith et al. 2001, Fensham and Fairfax 2005, Beringer et al. 2007). Woody thickening is considered to have contributed to recent changes in the fauna of tropical savannas (Tassicker et al. 2006). As woody plant cover and density increase, grass production typically declines dramatically (Scholes and Archer 1997). This leads to reduced seed available for granivorous birds. In Australia, woody thickening is likely to have contributed to the decline of the Golden-shouldered parrot (*Psephotus chrysopterygius*) on Cape York Peninsula (Garnett and Crowley 2002, Crowley et al. 2003).

Land use changes in Australian savannas – Water availability

European settlement has resulted in major changes in the distribution, availability and quality of fresh water across the landscape (SWQMS 2002). Under pastoralism, a vast number of artificial watering points – dams, bores, troughs – have been placed in the landscape (James et al. 1999, Pople and Page 2001, Montague-Drake 2004). Permanent and semi-permanent watering points become foci of grazing pressure (Andrew 1988, Thrash 1998) and this concentration of grazing alters vegetation composition (Thrash et al. 1993, Thrash 1998, Landsberg et al. 2003, Ludwig et al. 2004). The creation of artificial water sources allows some native species populations to expand their ranges and even persist in the areas where they were formerly sporadic or non-existent (Pople and Page 2001). The abundance and distribution of several species of birds have changed due to the provision of artificial watering points (James et al. 1999): some birds respond positively, increasing in abundance and range; others respond negatively, disappearing from areas that were formerly occupied (Reid and Fleming 1992). Water-dependent birds, such as granivores that require regular drinking water, have generally benefitted from the watering points (Reid and Fleming 1992).

Although the knowledge of each threatening process is increasing, and the decline of granivorous birds is recognised, there are still many gaps in our understanding. Increased knowledge of the drivers of granivorous bird declines in Australia, particularly in tropical savannas, may allow the implementation of more targeted management strategies to halt their decline (Franklin et al. 2005). The specific ecological requirements of granivorous birds in savanna ecosystems, how they use the landscape, which particular features in the landscape they are selecting for nesting, roosting or foraging and how they move around the landscape are unknown for most species. In particular, information on requirements is most crucial for species undergoing rapid and substantial decline. This project therefore focuses on the biology and conservation of the Black-throated finch (*Poephila cincta*), of which the southern subspecies has suffered a major range contraction.

The Black-throated finch, *Poephila cincta*

The Black-throated finch (*Poephila cincta*) is a granivorous bird endemic to north-eastern Australia (Zann 1976a) (Fig. 1.2) and occurs in open grassy woodlands. It belongs to the family Estrildidae and its genus, *Poephila*, includes two other species, the Masked finch (*P. personata*) and the Long-tailed finch (*P. acuticaudata*). Two subspecies of the Black-throated finch are recognized: the northern *P. cincta atropygialis*, which has a black rump, and the southern *P. cincta cincta*, which has a white rump (Higgins et al. 2006a, BTFRT

2007a, Buosi 2011). Historically, the southern subspecies ranged from the New England region of New South Wales (31ºS) through eastern Queensland to the region of the upper Burdekin catchment in north Queensland; while the northern subspecies is confined to Cape York Peninsula and the northern and western Gulf Plains, Queensland (BTFRT 2007a, Buosi 2011, Vanderduys et al. 2016).

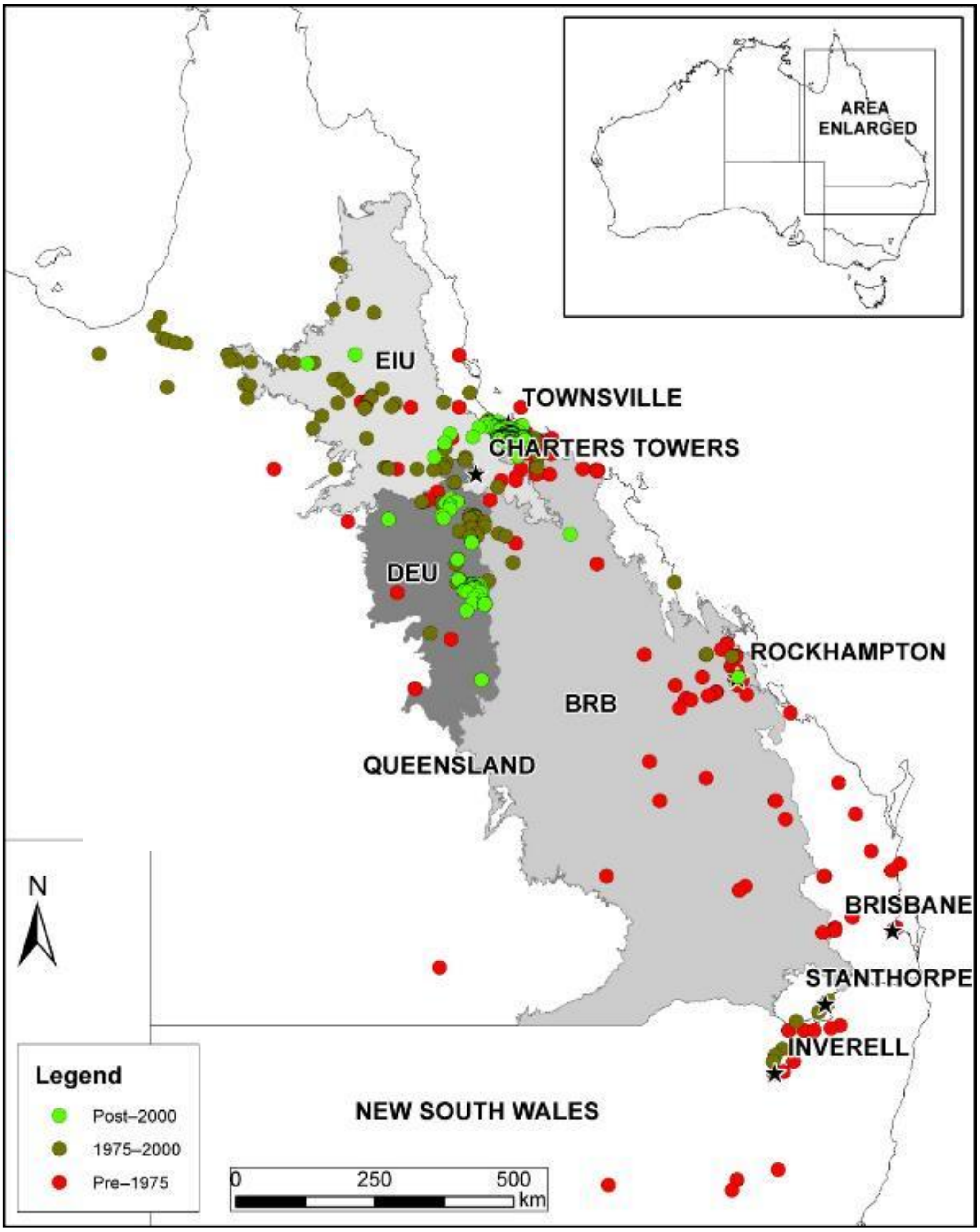

Fig. 1.2. Distribution of Black-throated finch southern subspecies records colour-coded by time period. Most relevant bioregions: BRB = Brigalow Belt, DEU = Desert uplands, EIU = Einasleigh Uplands (courtesy of Eric Vanderduys; see full map at (Vanderduys et al. 2016)).

Black-throated finch – conservation status

The Black-throated finch southern subspecies (*P. c. cincta*, hereafter referred to as BTF) has experienced an estimated 80% range contraction since the late 1970s (BTFRT 2007a, Buosi 2011), and the population is now restricted to the far northern part of its former range: Townsville Coastal Plain and inland areas of the Desert Uplands and Brigalow Belt Bioregions, south to about 23ºS (Fig. 1.2) (Buosi 2011, Vanderduys et al. 2016). There are no recent records of BTF from NSW where it may be extinct (Higgins et al. 2006a) and no more than nine records since 1980 in the southern part of its former range in southern Queensland (Fig. 1.2) (Vanderduys et al. 2016). BTF now appear to have two major strongholds – parts of the Townsville Coastal Plain in the northern Brigalow Belt Bioregion, and along the eastern edge of the Desert Uplands Bioregion (Vanderduys et al. 2016). Due to this range contraction and population decline, BTF is listed as 'Endangered' under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999 and NSW Threatened Species Conservation Act 1995, and 'Vulnerable' under the Queensland Nature Conservation Act (EPBC 1999, BTFRT 2007a, Buosi 2011) and in The Action Plan for Australian Birds 2010 (Garnett et al. 2011a). The IUCN classifies the BTF as 'Least Concern' at a species level, however BTF conservation status was 'Near Threatened' from 2004-2011 (BirdLife International 2012).

Black-throated finch – life history

BTF are predominantly granivorous, being recorded eating seed of 13 species, with *Urochloa mosambicensis* and *Digitaria ciliaris* being the most commonly observed grass species consumed in Townsville region (Mitchell 1996). *Setaria surgens, D. ciliaris*, *Stylosanthes humilis* and *Dactyloctenium radulans* were species observed to be contributing to the diet of the northern subspecies (Zann 1976a). Availability of seeds year-round is essential to BTF survival, and the demand for seed is especially high during the breeding season (Buosi 2011). BTF diet changes at the onset of the wet season, suggesting a reduction in availability of seeds as there is a shift from seeds to insects (Mitchell 1996). This observation on the Townsville Coastal Plain is consistent with the idea that birds experience a bottleneck in which there is a shortage of preferred food resources at the onset of wet season, as rains trigger germination of the remaining seed bank and the production of seeds by sprouting grass plants has not yet occurred (Franklin et al. 2005). To meet their energy demands in the season of declining availability of seeds, finches increased the breadth of their diet (Mitchell 1996). Unlike other granivores, such as the Nutmeg mannikin (*Lonchura punctulata*) or Chestnut-breasted mannikin (*L. castaneothorax*), BTF are ground foragers (Franklin et al. 2005), which presents particular issues in terms of food accessibility and availability.

The breeding season of the BTF appears to be dependent upon seed availability and therefore will differ from year to year according to rainfall patterns and land condition (Buosi 2011). In the Townsville region, the breeding season is believed to be 3-6 months after the onset of the wet season, and continues until seed availability declines (Buosi 2011). BTF build their nests in the outer branches of trees and shrubs, in close proximity to water and foraging dominates their activity budget during the breeding season (Isles 2007b, Buosi 2011). BTF usually nest in loose colonies and nests are used for both breeding and roosting (Buosi 2011). It is likely that BTF build their nests in areas of optimal habitat with respect to the availabilities of food, water and shelter (Buosi, 2009).

Little is known about BTF movements or home ranges (Higgins et al. 2006a). Daily movements of adults and fledglings are likely to increase after the breeding season, when fledglings are more mobile and resources near breeding sites are likely to be depleted (Buosi 2011). A study with non-marked birds showed the BTF average daily foraging movements were up to 400m during the breeding season (Isles 2007b). Another study with marked birds resighted individuals at locations 1.5 km apart in a single day, suggesting they might be capable of daily movements of 3 km (Mitchell 1996). Large scale movements in response to disturbances or resource needs might occur, but there is no information about such movements (Higgins et al. 2006a, Buosi 2011).

Black-throated finch – threats to conservation

Threats to BTF are believed to be generally similar to those that have been impacting granivorous birds of Australian tropical savannas generally: changes in vegetation structure, grazing intensity and clearing of woodlands for agriculture and pastoralism, altered fire regimes and weed invasions (BTFRT 2007a, Buosi 2011). Pastoralism, particularly when associated with heavy grazing regimes and/or woodland clearance, has had significant impacts on BTF foraging and nesting habitat (Buosi 2011). Recently, urban expansion has become an increasingly critical issue for some of the important known surviving BTF populations (BTFRT 2007a, Buosi 2011). Recent records suggest that BTF is increasingly becoming constrained to the contracting area of rural land on the Townsville Coastal Plain (Buosi 2011). Peri-urban areas surrounding Townsville not only lack BTF, but are unsuitable for other granivores such as Zebra finches (*Taeniopygia guttata*), Plum-headed finches (*Neochmia modesta*), Chestnut-breasted manikins (*Lonchura castaneothorax*) and Diamond doves (*Geopelia cuneata*), all of which are found in neighbouring rural areas (Whatmough 2010). It is not clear, however, what levels of disturbance from different land-uses and land-use practices most affect BTF, and to what degree.

The Brigalow Belt Bioregion is a key bioregion for the BTF (Buosi 2011) and encompasses the rapidly expanding city of Townsville. The most reliable and recent BTF records of sightings records are around Townsville and central Queensland (northern section of the Brigalow Belt), with the population existing on the Townsville Coastal Plain being easy to access for long term monitoring (Fig 1.2). Recent BTF sightings suggest that the largest known population is in central Queensland (Tang, L., pers. comm.), but that BTF population is threatened by mining activities (Vanderduys et al. 2016). Recent land clearing has been greatest in the Brigalow Belt Bioregion, which has lost over 50% of its native vegetation (SOEQ, 2007) and has the greatest number of endangered ecosystems of all bioregions in Queensland (Accad et al. 2013). Both main populations of BTF are under threat: Townsville Coastal Plain area is subject to urbanization, habitat loss, pressure from pastoralism, and invasive plants and animals; and the central Queensland area to mining and pastoralism. The survival of BTF populations in these areas is critical for long term conservation of the subspecies.

Ecological requirements and conservation actions

Understanding how animals are using available habitat, how they move through this habitat and identifying their requirements are prerequisites for successful conservation (Macnally 1990, Yen et al. 2011). Home ranges, movement patterns and the ways animals use the habitat – nest, forage or rest – are driven by abundance, availability and distribution of resources and habitat features, such as vegetation structure and composition (Jenkins 1981, Brandt and Cresswell 2008, Willems and Hill 2009, Beyer et al. 2010). For each of those life history activities birds will require specific environmental resources (Block and Brennan 1993): selection of nesting and foraging locations will be driven by factors such as the specific arrangement of food abundance, and might be different for each activity (Martin 1993b, a, Citta and Lindberg 2007, Barrientos et al. 2009). Therefore, identifying how animals are moving and using and selecting areas of the landscape is a major step for their conservation. For BTF the extensive range contraction that has occurred in the last 40 years means that conservation actions are paramount. The understanding of ecological needs of this species will improve conservation prospects by informing decisions about how to manage habitat, how much habitat is required and where conservation actions will be most effectively placed.

Thesis aims and objectives

The goal of this research was to enhance understanding of the ecological needs of the BTF in north eastern Australia. To address this goal, my aims were to:

1. Determine home range sizes and movement patterns, and
2. Understand habitat requirements

Aim 1: Determine home range sizes and movement patterns

Objective 1: Estimate home range sizes and habitat selection of BTF on Townsville Coastal Plain

Home range was investigated by radio-tracking individuals of BTF at two sites. In addition, I examined habitat selection to determine which habitats BTF were preferentially using. Habitat selection was based on a broad classification of the Australian vegetation, the Regional Ecosystems (RE). REs were overlaid with individual home ranges of BTF to determine which REs BTF were preferentially using (Chapter 2).

Objective 2: Determine local and long distance movements of BTF in savannas woodlands

BTF are believed to be sedentary birds but the food resources they use are patchily distributed. I examined daily and large-scale movement patterns by monitoring colour-banded and radio-tracked BTF (Chapter 2).

Aim 2. Understand habitat requirements

Objective 3: Identify habitat requirements by examining environmental variables, particularly vegetation structure and composition, important in determining BTF occurrence

Vegetation structure and composition are the main determinants of woodland bird communities (Tassicker et al. 2006, Johnson et al. 2007), so are important for understanding habitat requirements of BTF. Several locations on Townsville Coastal Plain were surveyed and BTF flock size or absence was related to vegetation structural and compositional characteristics (Chapter 3). Vegetation information for this objective was at a fine scale, with ground cover and woody plant cover variables collected in detail at each site. Vegetation surveys were conducted at each site to provide information about important vegetation features – structural and compositional variables - required by BTF within suitable REs.

Objective 4: Determine features of microhabitat associated with nesting sites of BTF

Understanding and protecting species nesting requirements is a crucial and often overlooked aspect of their conservation. I compared areas around trees used as substrate by BTF for nesting with randomly selected areas to determine which specific features in the landscape species determine nest selection (Chapter 4).

Objective 5: Determine relevant vegetation features in foraging patches used by the BTF

Animals tend to forage in patches of food in a way that optimises fitness considering factors such as prey-predator relationships, food abundance and vegetation structure and composition (Brown 1999, Fleishman et al. 2003). I compared areas where BTF were observed foraging with neighbouring areas (areas just adjacent to the foraging patch) and available habitat to determine which specific features in the landscape BTF were selecting to forage (Chapter 5).

Thesis outline and structure

The four data chapters of this thesis draw on the outcomes of two major components of field surveys differing in scope and research methods. The first data chapter presents results from actively catching, monitoring and following tagged birds, while the remaining data chapters examine at a fine scale the vegetation structure and composition where the species was seen. This contributes to Aims 1 and 2 (Fig. 1.3).

In this chapter (Chapter 1) I introduce underpinning theory and the background to this research, outlining the situation of granivorous animals in Australia. I also review how different land use changes are affecting the decline of granivores, focusing on a particular species and providing a rationale for the thesis.

Chapter 2 uses data from radio-tracking and banding activities in Townsville Coastal Plain to estimate home range sizes and movement patterns of BTF. This chapter explores home range, movements, behaviour (foraging, nesting, resting) and habitat selection (based on a broad Australian vegetation classification system, Regional Ecosystems). Chapter 3 relates BTF surveys (flock size) with vegetation surveys (ground and woody plant structure and composition) undertaken on Townsville Coastal Plain. Chapters 4 and 5 use data from the vegetation surveys as 'available habitat' and compare this with specific vegetation surveys in nesting locations and foraging patches, the 'used habitat'. Chapters 3-5 provide information at a much finer scale – within distinct REs – than habitat selection analysis conducted in Chapter 2.

Chapter 6 summarises all results in light of the aims and objectives. I then discuss implications of the research findings for conservation of the BTF and provide suggestions for future research.

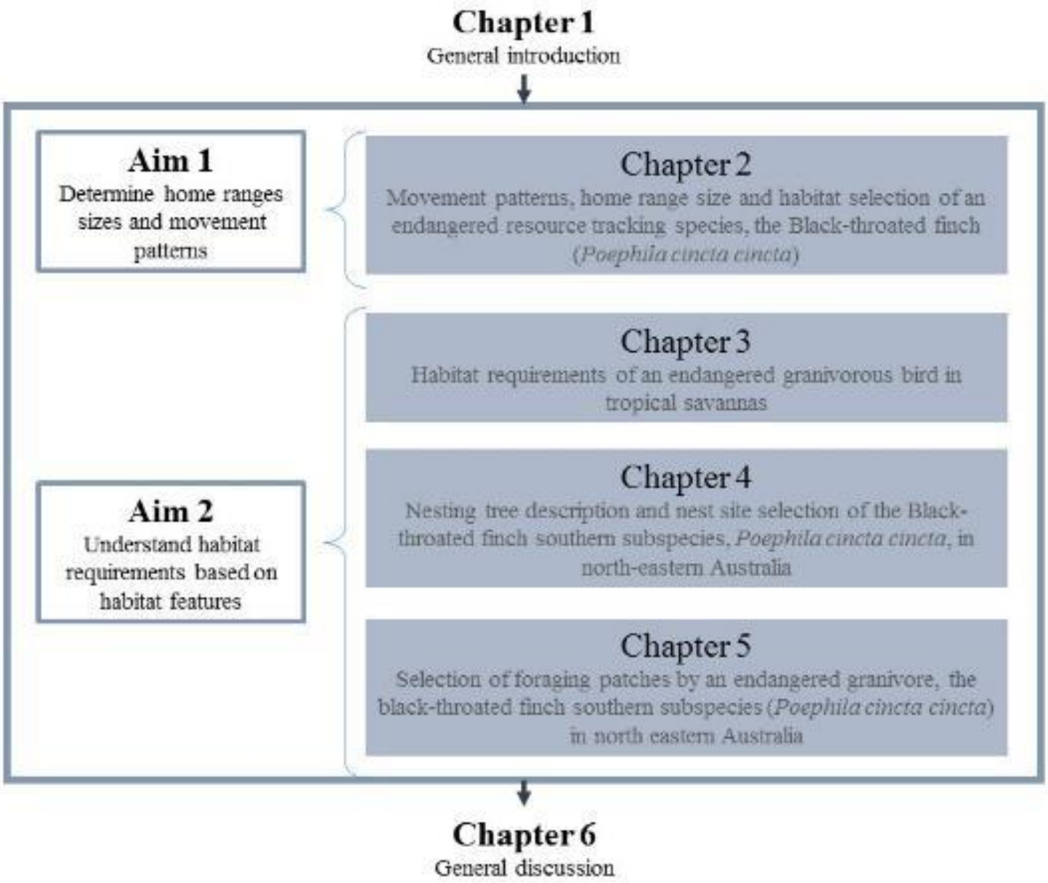

Fig. 1.3. Thesis structure.

Chapter 2

Movement patterns, home range size and habitat selection of an endangered resource tracking species, the Black-throated finch (*Poephila cincta cincta*)[1] in the dry season

Abstract

Understanding movement patterns and home ranges of species is important in conservation and management of threatened taxa. To address this knowledge gap for a range-restricted endangered bird, I estimated home range size, daily movement patterns and habitat use by a granivorous bird of northeast Australia, the Black-throated finch (*Poephila cincta cincta*; BTF) using radio-tracking and resighting of colour banded birds. Much of its habitat has been altered by pastoralism and agriculture. Little is known about basic aspects of its ecology including movement patterns and home range sizes. From 2011-2014, 102 BTF were colour-banded and 15 birds were radio-tracked. Home ranges were generated (calculated using kernel and Minimum Convex Polygons techniques). More than 50% of the resightings occurred within 200 m of the banding site (n = 51 out of 93 events) and within 100 days of capture. Mean home-range estimates with kernel (50%, 95% probability) and Minimum Convex Polygons were 10.6 ha, 50.8 ha and 46.3 ha, respectively. Home range size differed between sites but no seasonal differences were observed. BTF home ranges included four vegetation communities among eight available. Habitat selection was non-random at Site 1 (χ^2=373.41, df=42, p<0.001) and Site 2 (χ^2=1896.1, df=45, p<0.001); however the preferred habitats differed between the two sites. Some BTF moved further than expected on the basis of current knowledge, with three individuals being resighted over 15 km from the banding location. However, BTF generally maintain small home ranges over the short time-frames of this study. The occasional longer-distance movements may be related to resource bottleneck periods. Daily movement patterns differed between sites, which is likely linked to the fact that the sites differ in the spatial distribution of resources. The information about home range sizes and local movement of BTF from this study will be valuable for targeting effective management and conservation strategies for this endangered granivore.

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[1] This chapter was submitted as a paper to Plos One:
Rechetelo, J., Grice, A.C., Reside, A.E., Hardesty, B.D. and Moloney, J. 2015. Movement patterns, home range size and habitat selection of an endangered resource tracking species, the Black-throated finch (*Poephila cincta cincta*).

Introduction

Understanding how animals use landscapes to meet their demands for resources (food, water, and breeding habitat) and how those animals establish and use their home ranges is important for managing wildlife (Morrison et al. 1998, Kernohan et al. 2001, Willems and Hill 2009). Home range size, movement patterns and habitat use are driven by the abundance, availability and distribution of resources, as well as the structure of the landscapes in which they are distributed e.g., patchiness and connectivity; (Jenkins 1981, Brandt and Cresswell 2008, Willems and Hill 2009, Beyer et al. 2010). Understanding movement patterns and home range is particularly crucial when a taxon is threatened and dependent on targeted conservation measures. However this information is often lacking when taxa occur at low densities or are difficult to observe due to behaviour or other factors.

Granivorous birds are a common functional group in most terrestrial ecosystems, and particularly in savanna landscapes (Morton and Davies 1983, Franklin et al. 2000, Brandt and Cresswell 2008). Movement patterns for granivorous birds have been generally described as ‘extensive nomadic’ – irregular movements with destinations varying from year to year – and their home range as ‘very large’; this is because their main food resources (seeds), are patchily distributed in space and time due to the variability of rainfall and other environmental variables (Zann et al. 1995, Clarke 1997, Dean 1997, Brandt and Cresswell 2008). Nomadic animals may return to the same breeding area or may move to different areas, depending on environmental conditions (Dean 1997). However, under more predictable conditions, granivorous bird species are resident (Brandt and Cresswell 2008). In Australia, approximately 20% of all land bird species are granivorous but over 30% of all granivorous birds have declined in abundance and many are now threatened (Franklin et al. 2000, Garnett et al. 2009). Declines are more marked in tropical and subtropical savannas and the arid zone (Franklin et al. 2000). However, little is known about movement and home range of granivorous birds in Australia (Higgins et al. 2006a).

The Black-throated finch (*Poephila cincta*) is a granivorous bird of northern Australia that has suffered serious decline (BTFRT 2007a). The species was previously found in woodland habitats from north-east New South Wales to northern Queensland (Keast 1958, BirdLife International 2015b). The range of the BTF southern subspecies (*Poephila cincta cincta*) (BTF) has contracted by 80% since the 1970s and the sub-species is listed as threatened under Federal and State legislation (BTFRT 2007a, Department of the Environment 2015b). The population decline is likely to have been the result of habitat loss, primarily due to clearing and fragmentation of woodland for agriculture and impacts of domestic stock and invasive plants (Garnett and Crowley 2000b, BTFRT 2007a).

The important information about movement patterns, as well as habitat selection, that is required for effective management, is generally not available for the BTF (Higgins et al. 2006a, Department of the Environment 2015a). The aim of this research is to investigate the movements, home ranges and habitat selection of BTF, filling this knowledge gap to inform management and recovery plans for the sub-species.

Methods

Study Area

This study was conducted during 2012 - 2014 in the vicinity of Lake Ross, south of Townsville, Queensland, north eastern Australia (Fig. 2.1). This area is part of Townsville Coastal Plain and supports one of the most important extant populations of BTF (Garnett et al. 2011a). Most of Townsville Coastal Plain occurs at the northern boundary of the Brigalow Belt North Bioregion (Sattler and Williams 1999) and the study sites comprises a patchwork of public and private land (Chapter 3). The average annual rainfall at Townsville airport (ca.

18 km away) is 1,157 millimetres, most of which falls during the six months of the wet season, from November through April (Bureau of Meteorology 2015).

The vegetation in the vicinity of Lake Ross is mapped under 17 different vegetation types (Regional Ecosystems; RE) (BTFRT 2007a) but four of these cover more than 70% of the area. Vegetation is dominated by *Eucalyptus platyphylla*-*Corymbia clarksoniana* woodland; *E. crebra* or *E. paedoglauca* and *C. dallachiana* woodland; and *Melaleuca viridiflora* with occasional *M. argentea* woodland to open woodland (The State of Queensland 2014). The ground layer is usually grassy with common species including kangaroo grass (*Themeda triandra*), black speargrass (*Heteropogon contortus*), northern canegrass (*Mnesithea rottboellioides*) and giant speargrass (*H. triticeus*). Invasive grasses and shrubs are common throughout the area, occurring in scattered patches or extensive infestations. Common introduced species include Indian jujube (*Ziziphus mauritiana*), rubber vine (*Cryptostegia grandiflora*), stylos (*Stylosanthes* spp.), snakeweed (*Stachytarpheta jamaicensis*), guinea grass (*Megathyrsus maximus*), sabi grass (*Urochloa mosambicensis*), and grader grass (*Themeda quadrivalvis*) (Smith 2002, Grice et al. 2013).

Ethics statement

Ethics approvals for mist-netting, banding and radio-tracking was obtained from the Animal Ethics Committee N° 1693, James Cook University. Mist-netting, banding, color-banding and bander permits were obtained from the Australian Bird and Bat Banding Scheme (ABBBS) under authority N° 2876 and N° 3009. The Scientific Purposes Permit, N° WISP10390011, was issued by the Department of Environment and Resource Management under the legislation: S12 (E) Nature Conservation (Administration) Regulation 2006.

Mist Netting and banding

BTF were mist-netted between 2012 and 2014 at eight sites around watering points on private properties and land belonging to Townsville Water in the catchment area of Lake Ross, southeast of Townsville (Fig. 2.1). Granivorous birds need to drink water on a daily basis and it is common to see many such birds near water sources, which they visit regularly (Evans et al. 1985, Evans and Bougher 1987). Banding was attempted periodically throughout the year round but no birds were captured during the wet season. Banding sites were selected based on local information and the author's observations of water sources frequently used by the birds. Banding efforts began at sunrise and continued until about 11am (total of 1088.5 net hours); nets were closed earlier if the weather was hot, or if large numbers of other species were caught, and they were re-opened only after all birds had been processed. Birds were released where captured. Birds were uniquely banded with a numbered stainless steel and three darvic color bands (color code was according to the ABBBS). A subset of 15 of the 102 individuals banded were radio-tracked for home range estimation, with birds selected randomly relatively to the order in which they were captured; the number of BTF radio-tracked simultaneously varied from one to six (see Telemetry section).

Movement

Estimates of bird movements were based on recaptures and resightings of banded individuals. Specific searches for banded birds were conducted within a 2 km radius of the release locations. I calculated the distance each bird moved from the point of capture to each subsequent location. I recorded Universal Transverse Mercator (UTM) coordinates using a Global Positioning System (Model GPSmap62s, Garmin) for all captures and subsequent resightings of color banded birds. Resightings could arise from monitoring banding sites, water sources and surroundings, recaptures, during radio-tracking, or other fieldwork, such as vegetation surveys.

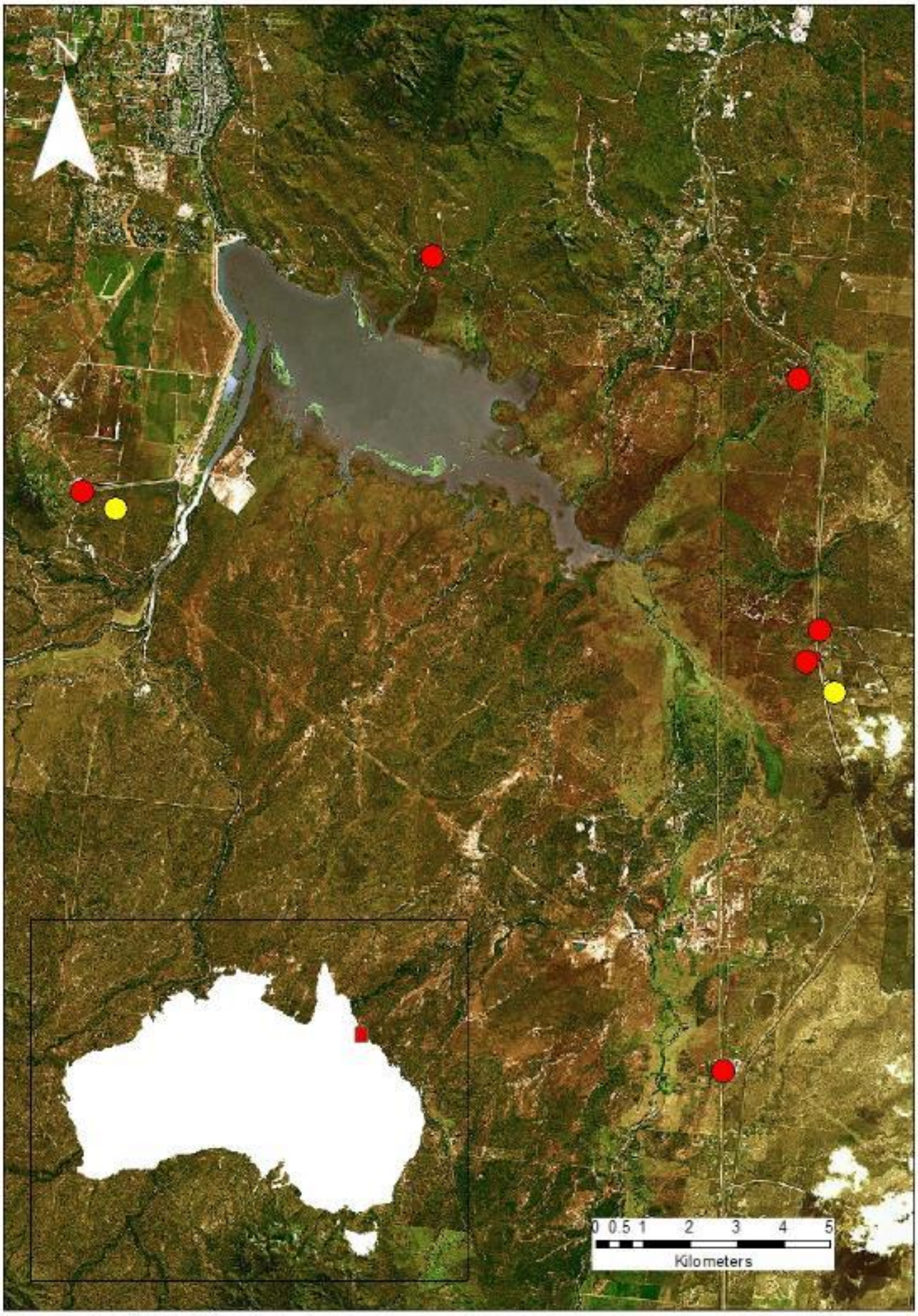

Fig. 2.1. Study area in Queensland, Australia: eight sites where banding was conducted; at two of them (yellow circles) radio tracking of BTF was also conducted. The grey area at the top of the map is Lake Ross. Parts of the urban area of Townsville are shown to its north.

Telemetry

Radio-telemetry was used at two of the eight banding locations; these were on land under local government jurisdiction with low intensity of land-use though there was some prescribed burning (Site 1) and on private cattle grazing property adjacent to a major road (Site 2; Fig. 2.1). Both areas comprised eucalypt woodland with a grassy understorey, but with a high proportion of introduced plant species (ground cover and understorey vegetation).

Seventeen birds were fitted with a 0.3g radio transmitter (Model A2414; 24 days battery life with operating frequency near150/151MHz; Advanced Telemetry Systems USA / Australia) placed in the scapular region. The feathers were trimmed to expose a small patch of skin and the tag was attached using cyanoacrylate glue and a piece of cotton fabric to increase surface area (Johnson et al. 1991, Anich et al. 2009, Smolinsky et al. 2013, Diemer et al. 2014). Transmitters were equivalent to no more than 3% of the bird's weight (= 14.1g, n= 163), which is within the range required to avoid any risk to the bird and behavioural changes (Millspaugh and Marzluff 2001, Withey et al. 2001).

After fitting the transmitter, each bird was held for 3 - 4 minutes, to test movement, and birds were then released at point of capture. Birds were tracked using an Ultra narrow band VHF receiver (Model VSR 042A) and three-element Yagi antenna (Sirtrack Ltd. 2015) by a single observer.

Radio transmitters were attached to captured BTF in 2012 (n = 1), 2013 (n = 10) and 2014 (n = 6) and the search for birds started the day after the capture in the area where each bird was mist-netted. When a bird was located, activity was recorded as foraging, nesting (including nest maintenance or roosting) or perching. After the bird had left a location, I recorded the UTM coordinates of that location using a GPS. If the bird was not observed and the observer's presence could flush the bird or interfere with its behaviour, I used the strength of the radio signal to estimate and mark the position (Vega Rivera et al. 2003, Ginter and

Desmond 2005). Individuals were tracked at different times of the day, from sunrise to sunset, five to 12 hours a day. Locations were recorded daily, at a minimum of 30 minute intervals, until the signal disappeared, was inconsistent, the transmitter was removed by the bird / fell off or the bird was found dead (unknown reason; n = 1). A transmitter was considered detached/removed if no evidence of predation was detected (Baldwin et al. 2010). The time of each observation was recorded and further classified into one of three categories: morning (sunrise to 10.00am), midday (10.01am to 14.00) and afternoon (14.01 to sunset).

Habitat selection

Habitat selection was analyzed by comparing actual habitat use (temporary home ranges) with that expected based on habitat availability (Brandt and Cresswell 2008). Available habitat was defined as the area within a circle centred on the trapping site and containing all locations recorded for the bird (Brandt and Cresswell 2008). The radius of the circle for both sites was determined by the distance between the catching point (water source) and the most distant point estimated by this home range. This distance was then used as a radius such that all the home ranges fell inside the circle.

RE descriptions from the Regional Ecosystems Description Database (REDD) (Accad et al. 2013, Protection 2014, The State of Queensland 2014) were used for available habitat. The non-remnant areas in RE layers were re-classified by delineating habitats based on satellite images from Google Earth (2014) in combination with direct observations in the field. The non-remnant areas were reclassified as cleared, mango plantation or re-growth. Habitat use categories included:

(1) REs – as per REDD. This classification takes into consideration vegetation communities of a particular bioregion that are consistently associated with a particular

combination of geology, landform and soil (Protection 2014, The State of Queensland 2014);

(2) cleared - areas with no trees, overgrazed (botanical composition, cover or erosion) or that had any major disturbance;

(3) mango plantations - monocultures of mangoes;

(4) re-growth - areas similar to the RE close by, with some disturbance in the past but with regrowth of local native species and some non-native species.

Data analyses

The utilization distribution (UD) (Worton 1989) estimated for the BTF represents the area the bird occupies over a short period of time (weeks to months) and not the area it will use over its entire life time. However, it will be referred to hereafter as home-range for individual birds.

Radio-tracking data were imported into a geographic information system (ArcGIS10, ESRI) and Geospatial Modelling Environment (GME) extension (Beyer 2012). To estimate home-range sizes and activity centres, I used the kernel method (Worton 1989) and Minimum Convex Polygons (MCP) for each individual. Kernel home range (KHR) is based on the probability of use derived from the number and spatial arrangement of locations and the relative amount of time an animal spends in a given area. Home range sizes were estimated at 50% (50%KDE; also referred to here as core areas) and 95% (95%KDE) probabilities (e.g., 95% kernel is defined as the area in which the animal is estimated to occur 95% of the time) (Worton 1989). Differences among estimated home-range sizes were compared between areas (Site 1 and Site 2) and between seasons (the period from November to January was classified as late dry season and May to October as early dry season). The wet season was considered to have started when a sequence of rain events occurred, usually in association with the southerly movement of the monsoon trough. Statistical differences in home ranges generated by kernel (95%KDE and 50%KDE) and MCP between areas and seasons were tested with Welch's t-test. Additionally, MCP home-range sizes were plotted against the number of locations for each bird; the stabilization of the curve, or its asymptote, indicates that the number of locations was sufficient (Harris et al. 1990).

All locations were biologically independent as birds could easily traverse their territories in less than the 30 minute interval between the recording locations (Barg et al. 2005). Statistical independence to eliminate autocorrelation, however, would require excessively long periods between successive times at which locations were recorded and would lead to the loss of biological data such as daily activity patterns and ecological information (Cushman et al. 2005). My approach sought to maximize the information gathered.

Individual BTF typically travelled in pairs or small groups and birds were observed to remain in the same flock most of the day during tracking periods. Thus, to analyse UD when birds were engaging in different activities (foraging, resting, and roosting), I used all locations (all tracked birds). The same procedure was followed when analysing utilization areas at different times of the day. It used χ^2 contingency tests for activity and day period. If the expected values did not meet the assumptions, I used Fisher's exact test. I used ANOVA to test for differences between flock size in different periods of the day and for different activities. Activity estimations had smaller sample sizes than 'time of the day' estimations as BTF activity could not always be determined without flushing the birds (see Telemetry section).

Daily distance travelled (m) and distance from the roosting nest (m) was calculated using UTM coordinates of the radio-tagged bird. Habitat selection was analysed through overlays of kernel density surfaces and vegetation data. Home ranges instead of bird locations were used to avoid problems associated with non-independence (Aebischer et al. 1993). Habitat selection was tested using two approaches: selection ratios for Habitat Selection Studies - design II (the habitats used are calculated for individual birds while available habitat was the same for the population) (Manly et al. 2002). Next, the approach of Calenge and Dufour (2006) was used to show the variability of habitat selection by performing an eigenanalysis of selection ratios (Calenge and Dufour 2006). Analyses were computed using the adehabitat package for R (R Core Team 2014).

Results

A total of 102 BTF were banded from 2012 to 2014; of those, I recaptured eight birds in nine recaptures/events. Of the banded individuals, I resighted 46 colour-banded individuals in 84 events (min = 1; max = 6; mean = 1.8 ± 1.3). A total of 49 individuals were recaptured or resighted in 93 events between 0 and 642 days after banding (mean = 143.04; median = 101). Approximately 50% of all resightings and recaptures occurred within 100 days (n = 46 events), while 15% of resightings (n = 13 events / 8 individuals) took place more than one year post initial capture.

More than 50% of the resightings occurred within 200 m of the banding site (n = 51 out of 93 events). Two individuals were resighted 5.0 and 6.2 km away respectively and three individuals were resighted over 15 km from the banding site (time elapsed between events was between 49 and 132 days; Appendix A, Table A.1). Five individuals were resighted in the same locale over 400 days after banding, and one individual was resighted in the same locale more than 600 days after banding. Among the individuals that moved over 15 km, one was recaptured/resighted three times in the same area it was banded (after 384, 412 and 560 days, respectively) and then resighted 16 km away 642 days after banding while the five resightings of one individual were all 16 km away from the banding area (Appendix A, Table A.1).

Out of the 17 tracked birds, one was killed by a predator and transmitters detached from two birds within a week. Birds were tracked for an average of 11.6 days (min = 1; max = 21; SD = 6.2; Table 2.1). The number of locations varied from 2-111 per individual (median = 47) and 1-11 per day, per individual (mean = 4.8; median = 5; SD = 2.4).

Home range sizes

Home range estimates were produced for 15 individuals. Eight birds had core areas (50%KDE) smaller than 10 ha (minimum and maximum number of locations were 24 and 111 respectively). Six birds had core areas of 10-15 ha and one individual had a core area greater than 29 ha (Table 2.1; Figs. 2.2 and 2.3). Mean home-range estimates with 50 and 95% probability and MCP were 10.6 ha (median = 9.1; min = 4.1; max = 29.3), 50.8 ha (median = 44.0; min = 25.2; max = 120.9) and 46.3 ha (median = 33.7; min = 22.9; max = 100), respectively.

Table 2.1. Kernel home-range estimates (ha) at 50% and 95% probability and Minimum convex polygon (MCP) for radio-tracked BTF. Seasons were defined as: LD (late dry season; November and January**) and ED (early dry season; May, July and September). Fates are defined as: LS (loss of signal), MO (mortality) and TD (transmitter detached). As: if home ranges reached asymptotes (Harris et al. 1990).

Site	BTF	Exposure days	Month	Year	Season	Locations	50%	95%	MCP	Fate	As
1	ANJ721	8	May	2013	ED	40	6.2	35.4	26.1	LS	No
	JOS722	7	May	2013	ED	41	4.1	25.7	23.0	LS	No
	ERC534	7	May	2013	ED	31	8.8	38.9	23.6	LS	Yes
	MAR726	15	July/August	2013	ED	75	6.8	36.5	29.8	TD	No
	LIL740	8	July/August	2013	ED	28	10.5	44.0	29.5	TD	Yes*
	CAR739	5	July/August	2013	ED	24	7.3	37.0	22.9	TD	No
	OWE737	1	July	2013	ED	2	-	-	-	MO	-
2	LEI699	12	December	2012	LD	51	5.1	25.2	25.7	TD	No
	JON750	1	September	2013	ED	2	-	-	-	TD	-
	REB751	15	September	2013	ED	42	15.6	96.6	95.1	TD	Yes
	JIM696	14	September	2013	ED	47	8.9	44.1	33.7	LS	No
	VIN856	14	January/February	2014	LD	80	10.2	50.4	41.3	LS	Yes
	BEA857	19	January/February	2014	LD	105	12.6	52.9	51.9	TD	No
	LUI854	20	January/February	2014	LD	103	29.3	120.9	100.1	TD	No
	GUI855	14	January/February	2014	LD	81	10.5	50.6	89.0	LS	Yes
	SOP858	21	January/February	2014	LD	111	9.1	39.7	39.4	TD	Yes
	LAS859	17	January/February	2014	LD	99	14.3	64.2	63.0	LS	Yes

* reach asymptote if outlier location is removed.

** In 2014 the effective wet season did not start until February so here January was still considered part of the dry season.

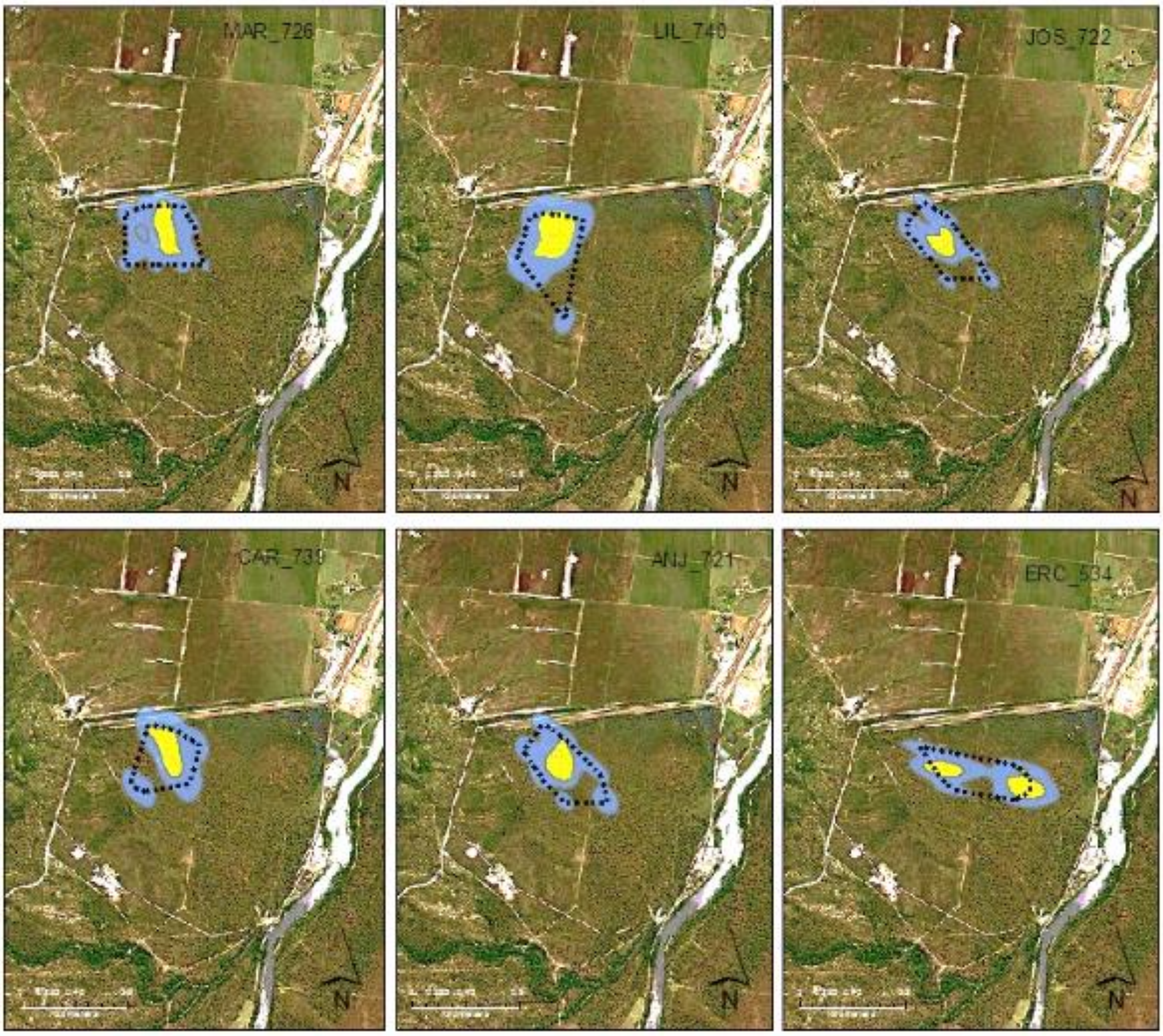

Fig. 2.2. Home ranges for 6 individuals of BTF *Poephila cincta cincta* calculated with 95%KDE (blue fill), 50%KDE (yellow fill) and MCP (dashed line) at Site 1, south Townsville.

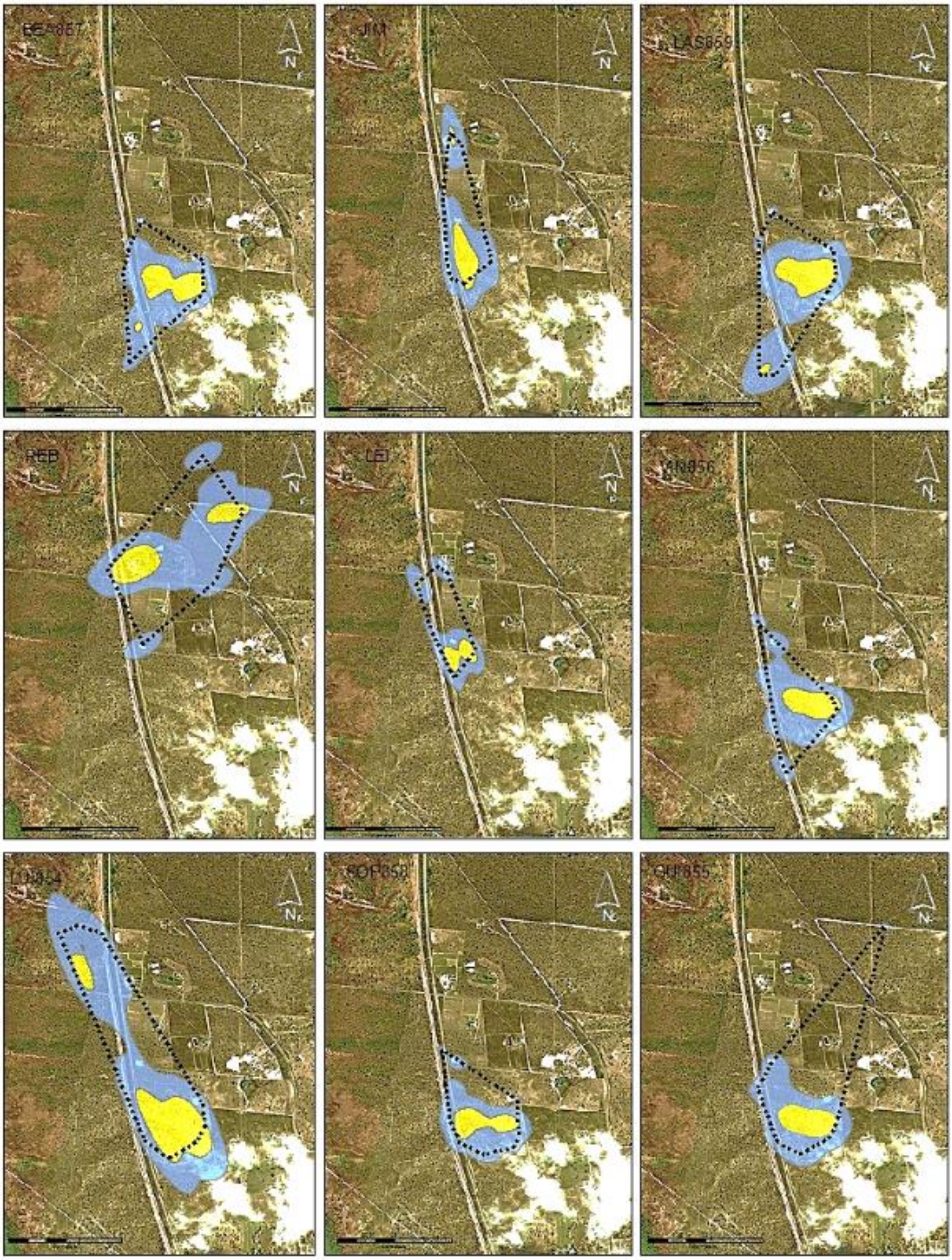

Fig. 2.3. Home ranges for 9 individuals of BTF *Poephila cincta cincta* calculated with 95%KDE (blue fill), 50%KDE (yellow fill) and MCP (dashed line) at Site 2, south Townsville.

Range size vs number of locations was plotted for 15 birds. Six birds clearly reach the asymptote (31, 42, 80, 81, 99 and 111 locations); however, some birds, even some with more than 50 locations, did not reach the asymptote (51, 75, 103 and 105 locations). Most of the asymptotes showed a stepwise arrangement (Appendix A; Figs. A.1, A.2 and A.3). Because of the inconsistency of BTF reaching asymptotes, all 15 birds were used in further analysis.

Home ranges MCP ($t_{2,8.3} = 3.58$, $P = 0.006$), 95% KDE ($t_{2,8.9} = 2.36$, $P = 0.004$) and core areas ($t_{2,10.2} = 2.24$, $P = 0.05$) were significantly smaller at site 1 than at site 2 (Fig. 2.4). Home ranges in the early dry season did not differ from home ranges in the late dry season, either for kernel home ranges (95%KDE: $t_{2,10.7} = 0.93$, $P = 0.37$; 50%KDE: $t_{2,8.1} = 1.4$, $P = 0.19$) or MCP home ranges (MCP: $t_{2,12.2} = 1.72$, $P = 0.11$; Fig. 2.5).

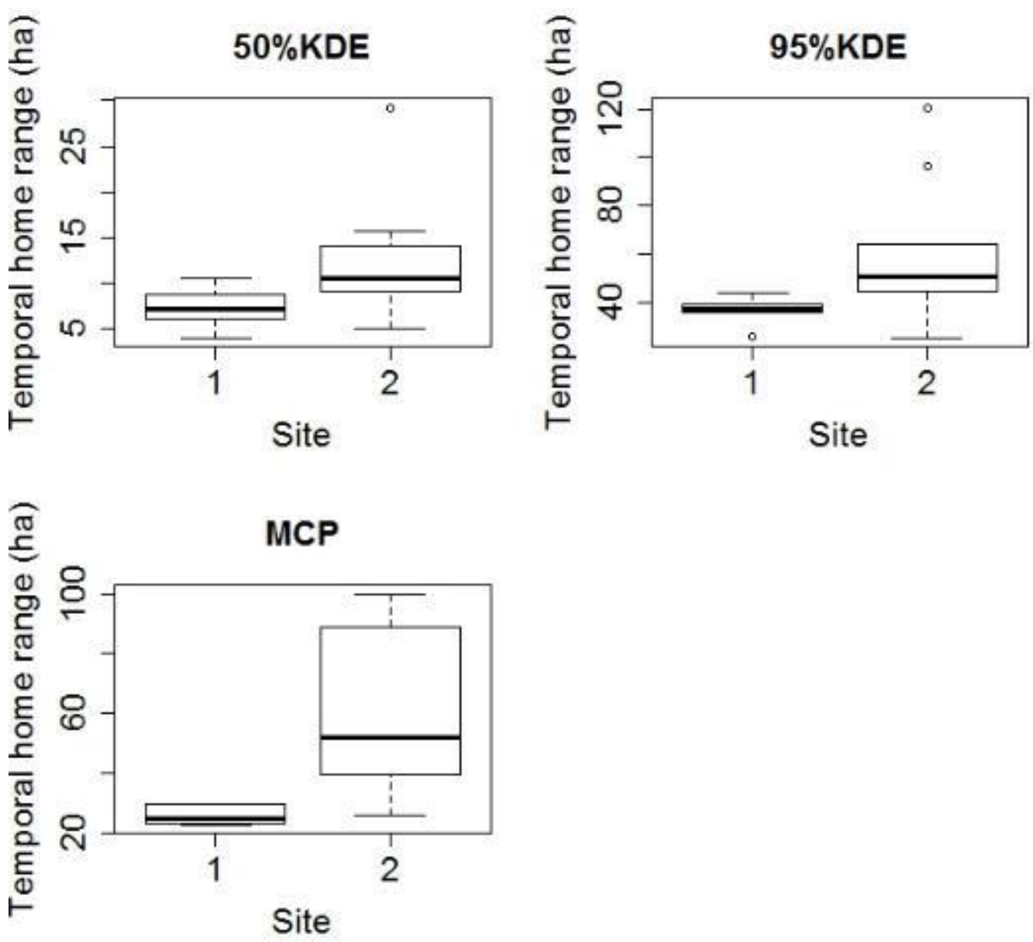

Fig. 2.4. Box-plot for temporal home ranges calculated for 15 BTF in Sites 1 and 2: 50%KDE, 95%KDE and MCP.

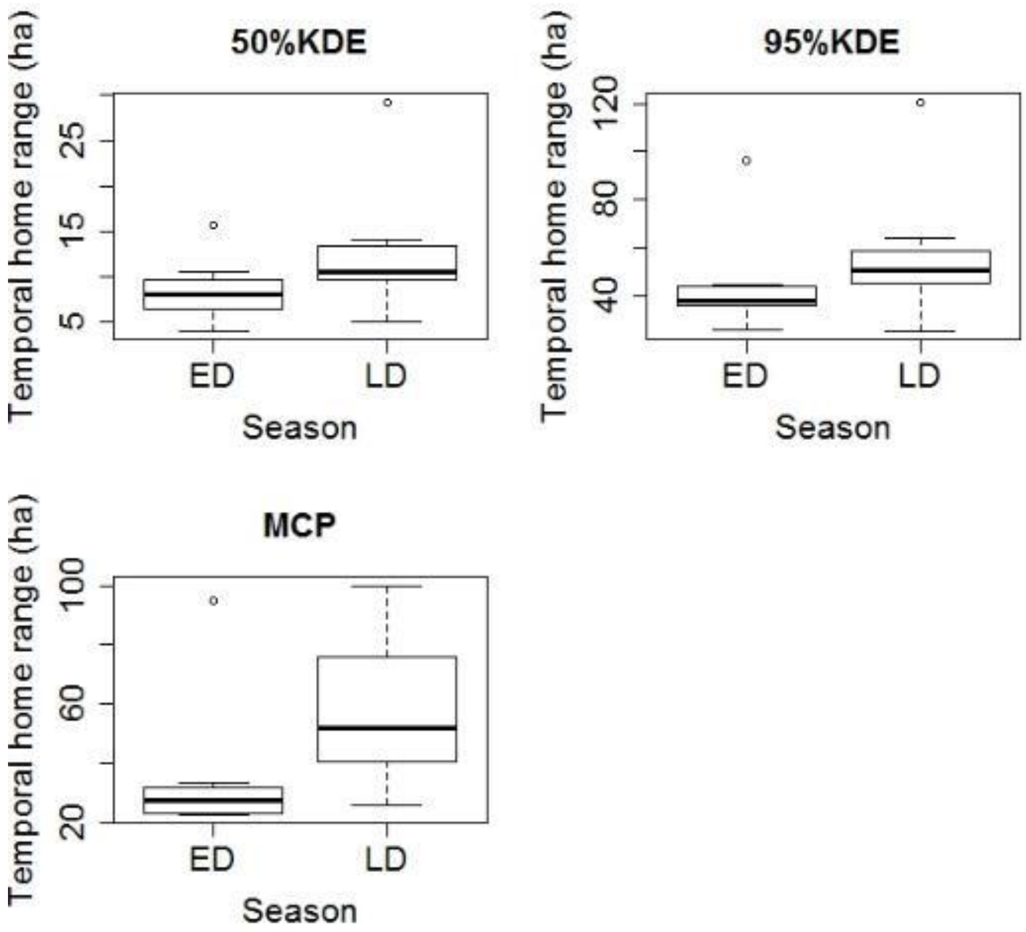

Fig. 2.5. Box plot for temporal home ranges of BTF in early dry season (ED) and late dry season (LD) calculated for: 50%KDE, 95%KDE and MCP.

Activity and time of the day

Foraging areas calculated for the flocks were of similar sizes at the two sites (although MCP for Site 2 was three times larger than for Site 1); the resting area was smaller than the foraging area for Site 1, however there was a 27.1 ha (95%KDE) and 6.27 ha overlap (50%KDE) between the two areas (Table 2.2; Fig. 2.6), which means 64% of the foraging area was also used for resting. At Site 2 the resting area was greater than the foraging area with 44.07 ha overlapping (95%KDE; Table 2.2; Fig. 2.6) between the two activity areas, which means 89.5% of the foraging area was also used for resting. The most pronounced difference between the sites was that birds at Site 2 used a larger area for roosting (Fig. 2.6).

Areas used by the flock varied with the time of the day: at both sites the area used by the birds is smaller in the middle of the day than in the morning or afternoon; at Site 1 it was greatest in the afternoon and at Site 2 it was greatest in the morning (Table 2.3; Fig. 2.7). BTF activities were not independent at different times of the day (morning, midday and afternoon) at Site 1 ($p = 0.02$; Appendix A; Fig. A.4) or Site 2 ($p < 0.001$; Appendix A; Fig.A.5). BTF flock size (521 flock size recordings) was variable (mean = 15.2; median = 14; min = 1 and max = 50) and differed significantly with time of day ($F_{2,518} = 41.89$; $p < 0.001$; Appendix A; Fig. A.6) and activity ($F_{2,505} = 35.55$, $p < 0.001$). BTF flocks were significantly larger at Site 2 ($t_{2,400} = 13.7$, $P < 0.000$; Site 1: 7.6 ± 5.1; Site 2: 17.5 ± 10.7).

At Site 1 birds remained in the core area all day whilst at Site 2 they established separate roosting and feeding areas. Six out of nine birds radio-tracked at Site 2 had regular daily activities, travelling from a specific roosting site to the foraging area, where they would spend the day. Usually the flock on the foraging area was larger (Appendix A; Figs. A.7 and A.8).

Table 2.2. Home range estimates (ha) for activity areas – foraging, resting and roosting (including nesting maintenance) for BTF in south Townsville, Queensland. Sample sizes are smaller because flocks were not always approached to avoid flushing them.

	Site 1			**Site 2**		
	Foraging (n=55)	Resting (n=94)	Roosting (n=5)	Foraging (n=232)	Resting (n=194)	Roosting (n=39)
50%KDE	8.02	5.1	-	8.31	11.18	38.95
95%KDE	42.29	35.36	-	49.24	65.81	144.95
MCP	43	40.94	2.06	131.21	217.11	80.42

Table 2.3. Home range estimates (ha) for different day periods – morning (sunrise to 10am), midday (10.01 to 14.00) and afternoon (14.01 to sunset) - for BTF near Townsville, Queensland.

	Site 1			**Site 2**		
	Morning (n=77)	Midday (n=95)	Afternoon (n=69)	Morning (n=295)	Midday (n=161)	Afternoon (n=265)
50%KDE	10.09	5.24	14.07	22.79	16.52	14.94
95%KDE	52.47	36.53	60.81	142.96	96.37	100.83
MCP	65.74	48.19	43.94	310.73	167.29	236.99

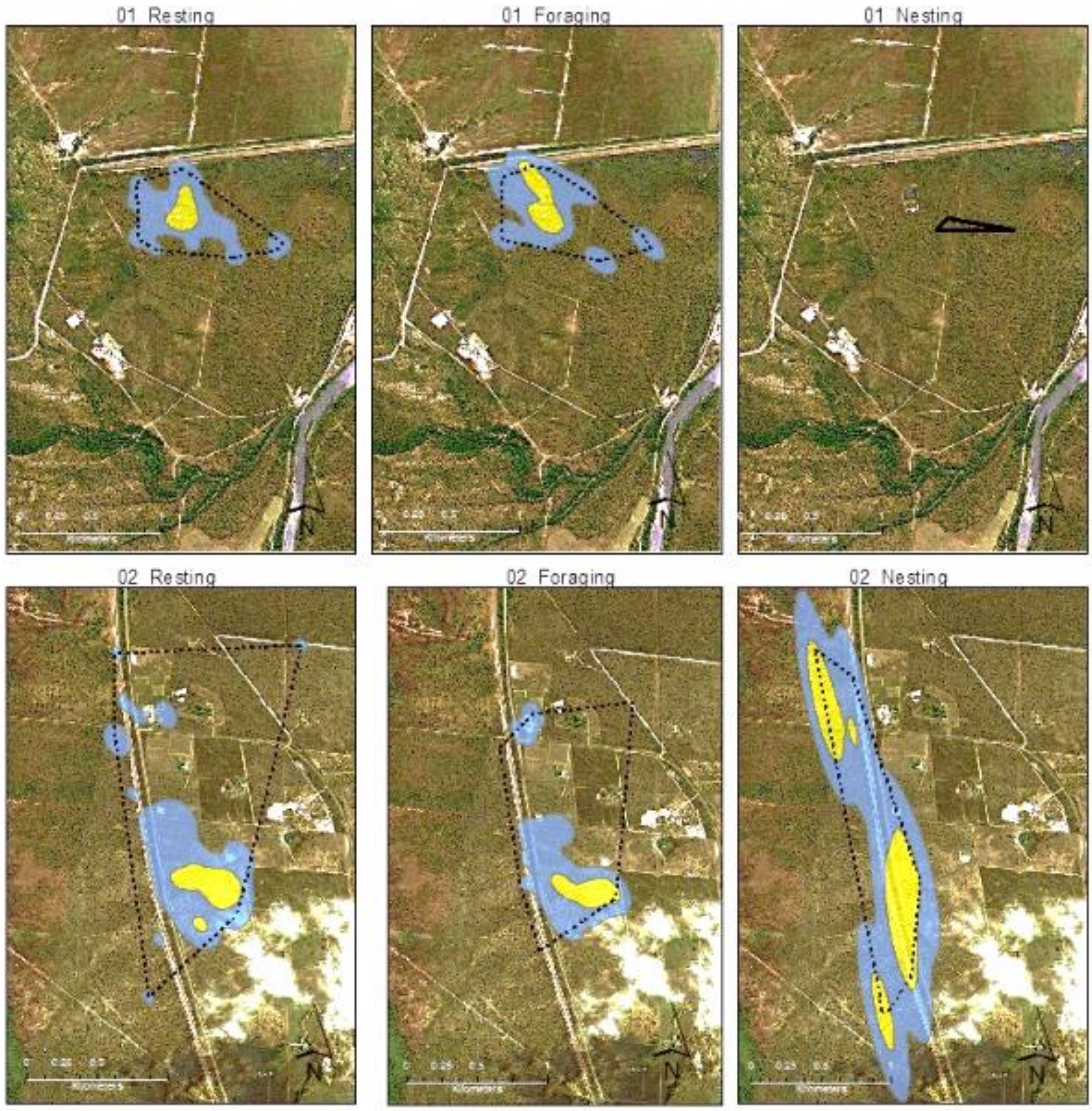

Fig. 2.6. Home ranges for the flock of BTF *Poephila cincta cincta* calculated with 95%KDE (blue fill), 50%KDE (yellow fill) and MCP (dashed line) at Sites 1 (top) and 2 (bottom) for different activities (resting, foraging or roosting), near Townsville, eastern Queensland. Home ranges in this analysis are for multiple individuals (flock) tracked at the same site.

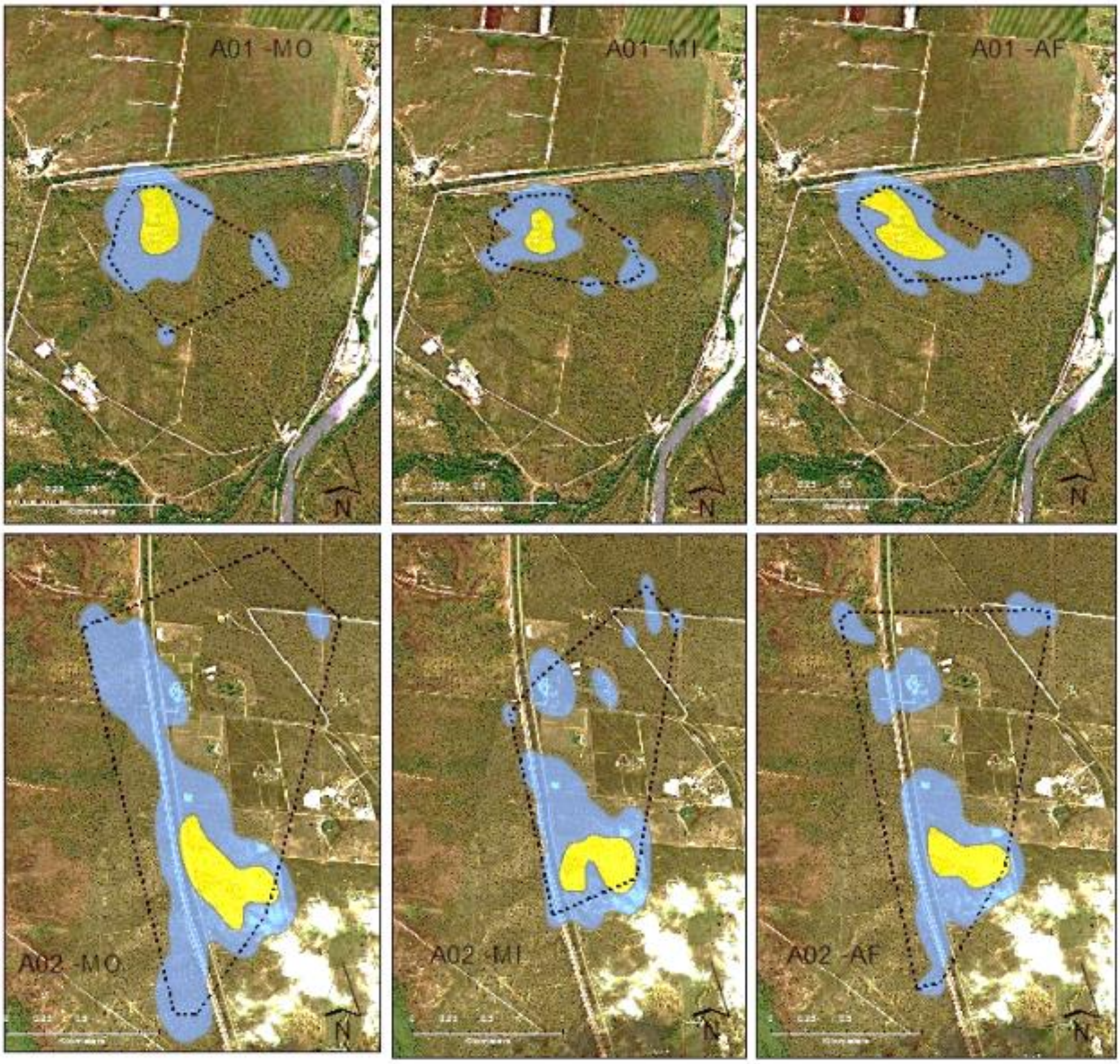

Fig. 2.7. Home ranges for the flock of BTF *Poephila cincta cincta* calculated with 95%KDE (blue fill), 50%KDE (yellow fill) and MCP (dashed line) at Sites 1 (top) and 2 (bottom) for different time of the day (MO= morning, MI= midday and AF= afternoon), near Townsville, eastern Queensland.

Habitat selection

BTF home ranges coincided with four vegetation types among eight types available at Site 1: *Eucalyptus platyphylla* woodland (RE 11.3.35), *Melaleuca* woodland (RE 11.3.12), re-growth and cleared areas (see Table 2.5 for RE descriptions). At Site 2, home ranges coincided with five habitat types among six vegetation types available: *Eucalyptus crebra* woodland (RE 11.3.30), *Eucalyptus platyphylla* woodland (RE 11.3.35), mango plantations, re-growth and cleared areas. Habitat selection was non-random at Site 1 (χ^2=373.41, df=42, p<0.001; Fig. 2.8), with birds preferentially using the *Eucalyptus platyphylla* woodland (RE 11.3.35), re-growth and *Melaleuca* woodland (RE11.3.12). Vegetation community selection was also different from random at Site 2 (χ^2=1896.1, df=45, p<0.001; Fig. 2.8) and the preferentially used communities were *Eucalyptus crebra* woodland (RE 11.3.30) and mango plantations.

Table 2.4. Vegetation types overlaid with BTF home ranges in Site 1 and 2.

Vegetation type	RE	Description
***Eucalyptus platyphylla* woodland**	11.3.35	*Eucalyptus platyphylla*, *Corymbia clarksoniana* woodland with *C. tessellaris* occurring in some areas. A low tree layer of species such as *Planchonia careya*, *Pandanus spiralis*, *Melaleuca viridiflora* or *M. nervosa* is often present. The ground layer is usually grassy with common species including *Themeda triandra*, *Heteropogon contortus*, *Mnesithea rottboellioides* and *Bothriochloa decipiens*.
Melaleuca woodland	11.3.12	*Melaleuca viridiflora* with occasional *M. argentea*, *M. dealbata* woodland to open woodland. Occasional midstratum of *Grevillea pteridifolia* and *Acacia leptocarpa*. Ground layer of perennial grasses such as *T. triandra*, *Elionurus citreus*, *Ectrosia leporina*, *Eriachne rara*, *Eremochloa bimaculata*, *Thaumastochloa pubescens*, *Eragrostis brownii* and *Ischaemum australe*.
***Eucalyptus crebra* woodland**	11.3.30	*Eucalyptus crebra* or *E. paedoglauca* and *Corymbia dallachiana* woodland. Forms an open woodland to open forest in places. Has a grassy ground layer of *H. contortus*, *B. bladhii*, *T. triandra*, *Sehima nervosum* and Enneapogon spp.

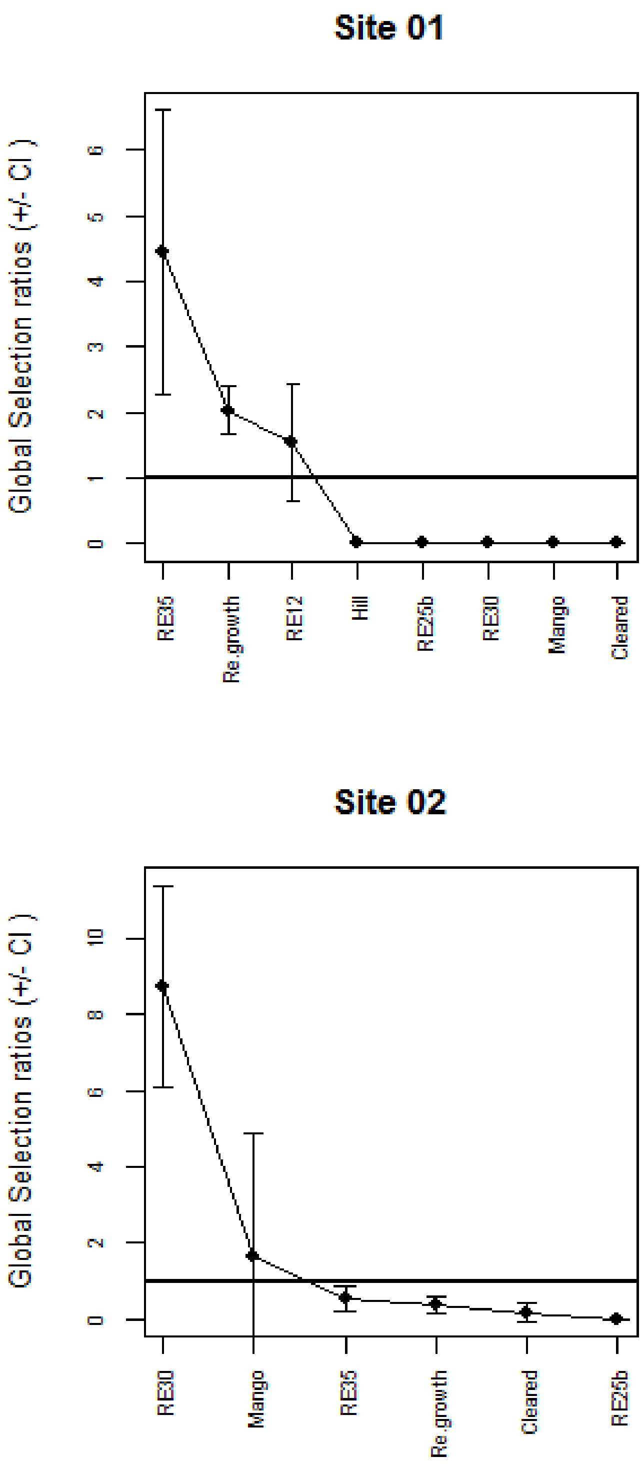

Fig. 2.8. Global Manly Selection ratios ± Confidence Intervals (CI) of the vegetation types analysed in Site 1 and Site 2 for BTF in south Townsville, eastern Australia. Mean selectivity rate of each habitat type is represented by black dots (•). Habitats within Global Selection ratios in the interval 0-1 are considered to be avoided by the birds, while habitats larger than one (horizontal line) are considered positively selected.

Eigenanalysis of selection ratios showed that four out of six individuals chose regrowth areas at Site 1 (re-growth areas; Fig. 2.9; red dashed circles); one chose *Melaleuca* woodland (RE 11.3.12; green dashed circle) and one *Eucalyptus platyphylla* woodland (RE 11.3.35; blue dashed circles). At Site 2, six individuals out of nine chose one type of habitat in Site 2, *Eucalyptus crebra* woodland (RE 11.3.30; Fig. 2.10; red dashed circles). Both results indicate variability in habitat selection, with individual BTF displaying different patterns of preference within a site and between sites.

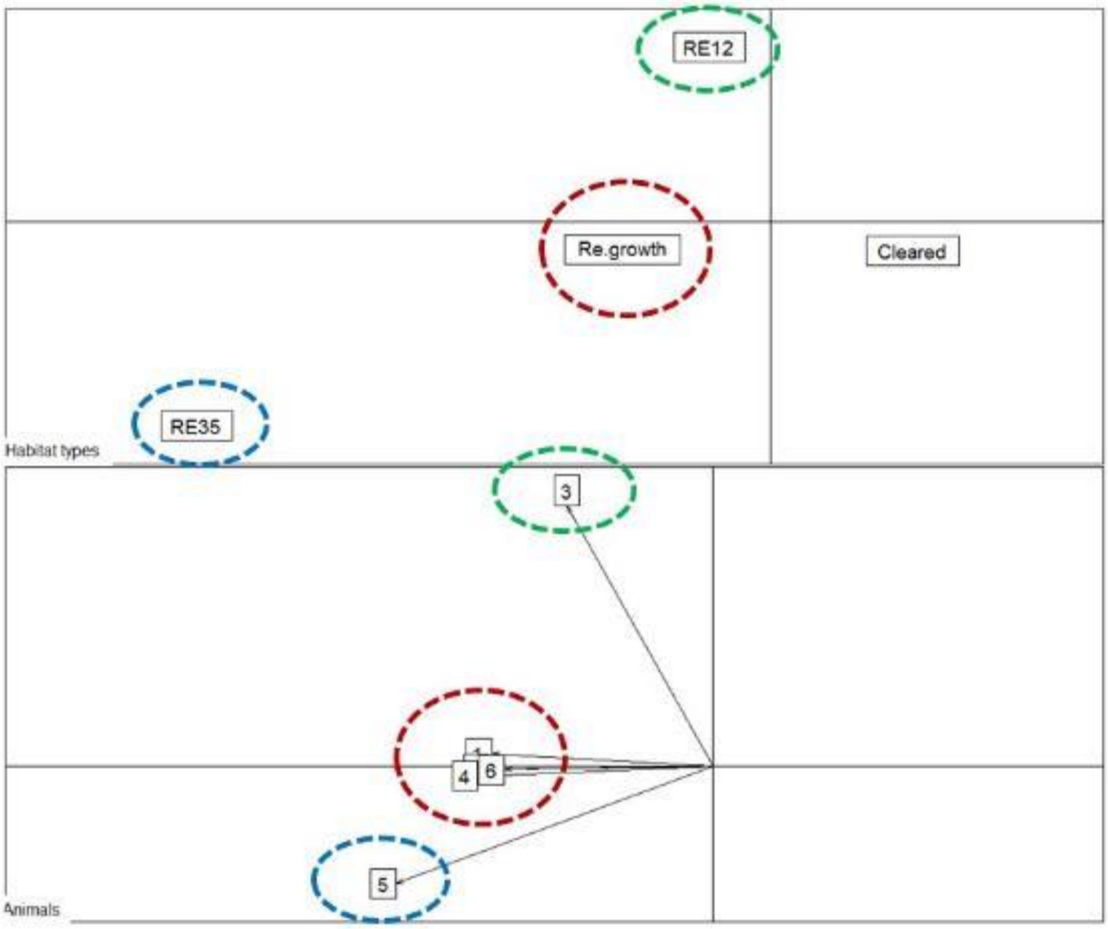

Fig 2.9. Results of the eigenanalysis of selection ratios to evaluate habitat selection for BTF in Site 1. The top figure shows the habitat types. The bottom figure shows habitat preference of each individual monitored.

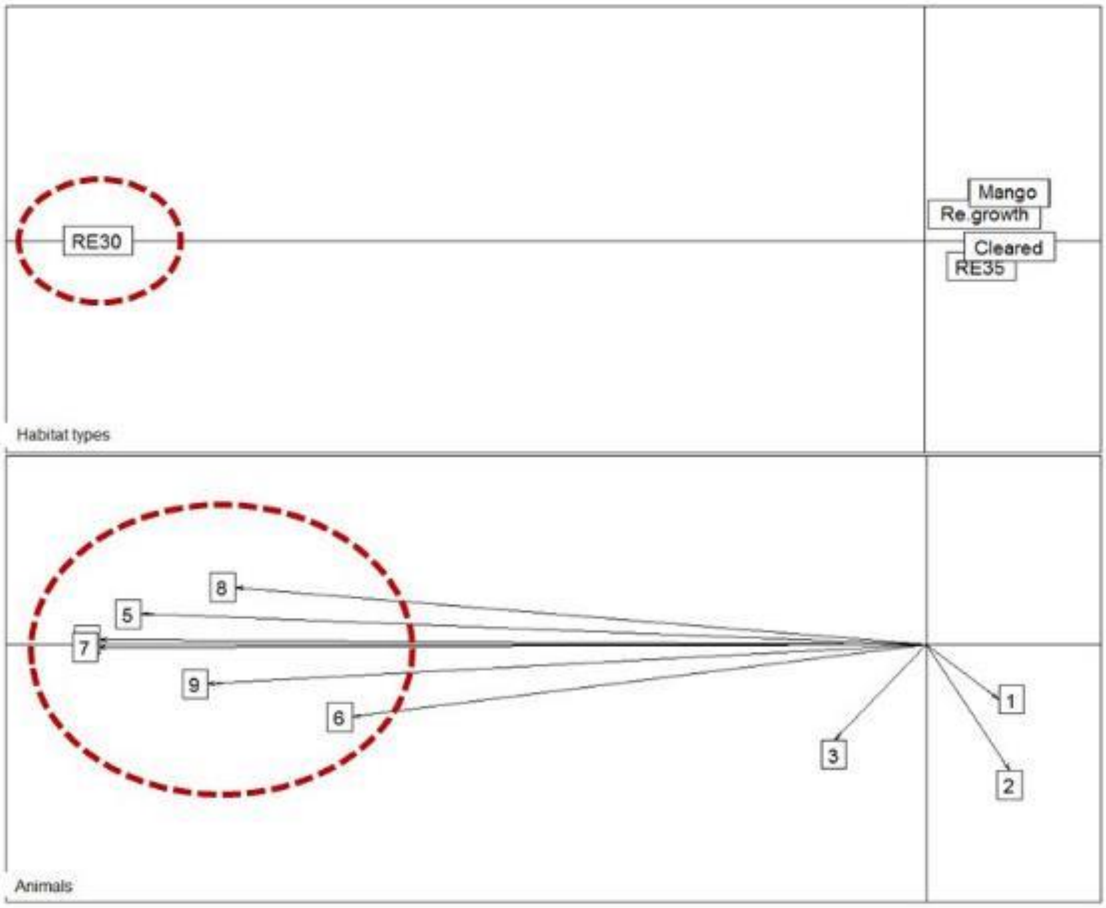

Fig 2.10. Results of the eigenanalysis of selection ratios to evaluate habitat selection for BTF in Site 2, South Townsville. The top figure shows the habitat types. The bottom figure shows habitat preference of each individual monitored.

Discussion

Home range sizes

BTF maintain small home ranges (ranging from 25.15 to 120.88 ha) over short time scales (e.g. within seasons). Previous studies suggested BTF foraging areas ranged from 2.3 to 4.4 ha (Isles 2007b), with banded birds located 2.5 km away from banding sites (Mitchell 1996). Some individuals were sighted over 15 km away from the banding site, indicating some long distance movements. Movement patterns for granivorous birds have been described as ‘extensive nomadic movements’ (Davies 1984), a feature attributed to the fact that their main feeding resources – seeds – are patchily distributed in space and time due to the variability of rainfall (Clarke 1997, Dean 1997, Brandt and Cresswell 2008). Despite these descriptions, BTF on Townsville Coastal Plain have movements that can be more accurately described as resident and sedentary (Higgins et al. 2006a).

BTF presented a fixed home range in the short time frames over which individuals were tracked. Species with fixed home ranges move repeatedly within a small area relative to their ability to travel and this type of movement is used to explore available resources (Gordon 2000). Similar findings were observed for other seed eaters such as Emberizid sparrows (Cassin's sparrow *Aimophila cassinii*, Brewer's sparrow *Spizella breweri*, Vesper sparrow *Pooecetes gramineus*, Savannah sparrow *Passerculus sandwichensis*, Grasshopper sparrow *Ammodramus savannarum* and Baird's sparrow *Ammodramus bairdii*), which had fixed home ranges during winter (Gordon 2000). Gouldian finches in Australia also appear to have fixed home range in the dry season when they remain in close proximity (2 km or less) to water sources and restrict their movements to nearby feeding areas (O'Malley and Team 2006, Lewis 2007). Animals can often adjust their movement strategies according to environmental conditions (Gordon 2000); a behavioral plasticity that is particularly important in unpredictable environments. It is likely that the BTF in this study were adjusting the area used in response to resource availability, consistent with the idea that granivorous bird species are resident in more predictable conditions (Brandt and Cresswell 2008).

Mean home-range estimate for BTF was 50.8 ha (95%KDE) and ranged from 25.15 ha to 120.88 ha. Individual home range sizes for Gouldian finches are much greater than those of BTF, often exceeding 2000 ha, but these data were based only on resightings data and over a 4-year period) (Lewis 2007; Maute 2011). Home ranges recorded for other granivorous birds were within the range observed for BTF: Yellowhammer (*Emberiza citronella*) home range was 149 ha and for Chaffinch (*Fringilla coeleb*) in farmland areas in Scotland it was 51 ha (Calladine et al. 2003). BTF populations were much more restricted to their home ranges over small temporal scales, but their home ranges moved incrementally over time. These increments occur as the birds look for resources in new areas, leading to larger home ranges when examined over a year. BTF could have similar movement patterns to those described for Zebra finches (extended local excursion, extra-home-range wanderings and large-scale movements; Zann 1996), so further investigation over longer time periods is required.

BTF home ranges differed between sites but not between seasons. There was no significant difference in BTF home ranges between early dry season and late dry season in this study. However, there were some differences, suggesting that more specific studies investigating seasonal home ranges may reveal significant patterns. Home ranges 100% MCP for Rock firefinches (*Lagonosticta sanguinodorsalis*) (Brandt 2007) were larger during the dry season than the wet season. Rock firefinches, like BTF, feed on grass seeds on the ground, which is likely to be a more stable food supply as seeds remain on the ground longer than they are held on the plant. This might enable birds to remain in the same area throughout the year (Brandt 2007). Further studies should investigate home ranges of BTF in the wet season. BTF home ranges at Site 2 were larger than those at Site 1. There are two possible reasons for this: 1) Site 2 was more fragmented and birds had to move further between roosting and foraging habitats in order to meet their requirements or 2) at Site 1 home ranges might be underestimated because only a few locations were collected per bird.

Movements

In this study, occasional long-distance movements (> 15 km) were observed. This movement, however, was based on resightings of banded birds recorded at widely separated locations, so the movements recorded here might not represent a single journey. Occasional long distance movements (> 10 km) have been described for several grass finches in Australia based on ABBBS records. Although most of the recaptured birds were within 10 km of the banding site, records of 10 - 49 km were recorded for Double-barred finch (*Taeniopygia bichenovii*), Red-browed finch (*Neochmia temporalis*) and the other species of *Poephila*: Long-tailed finch (*Poephila acuticauda*) and Masked finch (*Poephila personata*) (Higgins et al. 2006a). Long distance movements for Gouldian finch (> 25 km) have also occasionally been reported (Maute 2011). Long distance movements may be related to the seed bottleneck period, which occurs at the beginning of the wet season, when rainfall results in germination of much of the remaining accessible seed and seed production has not yet begun (Garnett and Crowley 1995, Crowley and Garnett 1999, Crowley et al. 2003, Franklin et al. 2005). Birds struggling to find resources undertake long distance movements. Zebra finches (*Taeniopygia guttata*) have been observed undertaking large-scale dispersive moments outside home ranges in prolonged dry periods (Zann 1996, Higgins et al. 2006a). Prolonged dry seasons may have influenced the long distance movements recorded for three BTF (resightings in August, June/October and February 2014 – the last was still dry, two months after the usual start of the wet season). BTF can and do move relatively long distances but it is not known how often or under what circumstances this occurs. It is important to understand the role these longer distance movements play in the overall ecology of BTF, if, for instance, they are being driven by naturally fluctuating ecological conditions or environmental trends that are detrimental for them.

Despite the difficulties in finding and capturing BTF, the colour banding technique was useful to show long distance movements of this species for the first time. Most of the information came from the resightings and provided useful information on some aspects of BTF longevity. Recapture rates are often insufficient to infer longevity and population structure; as found in this study and for Gouldian finches (Lewis 2007). In a study conducted on Townsville Coastal Plain, there were resightings with only six BTF from 82 banded individuals recorded (Mitchell 1996). The recapture/resighting rate in this study can be considered high when compared with many studies of Australian birds (Higgins et al. 2006a) thus providing unique ecological information for this species.

More than half of all resightings were within 200 m of where the birds were caught, which was at a water source. This habit of remaining close to the water sources where they were captured was also observed in Gouldian finches (Lewis 2007). Lewis (Lewis 2007) related this pattern to the breeding status of the birds, which would avoid flying away from the nesting site. Twelve nests of the 15 radio-tracked BTF were found, but nest height made it difficult for us to check if the nests were being used exclusively for roosting or also for breeding. The movements of Zebra finches (*Taeniopygia guttata*) are largely driven by rainfall patterns and water availability (Higgins et al. 2006a). This also appears to be true for BTF as the birds are more likely to remain close to the water sources in the dry season. This picture is supported by field observations in the wet seasons, where resightings of BTF were sporadic and no birds were caught during mist netting.

This study was the first to report daily movements of BTF as well as movement patterns between roosting and foraging sites. Daily movement patterns of BTF are likely to be influenced by the local environment, in particular the pattern of resource distribution (Ginter and Desmond 2005), as daily movements differed between the two study sites. BTF at Site 2 had designated roosting areas, where they spent the night and early hours of the next day for nest maintenance. The birds flew to the roosting areas late in the afternoon and returned to foraging areas next morning. This was not observed at Site 1, where four nests were in the same area in which foraging activities occurred while one bird had a nest approximately 800 m from the foraging area. Gouldian finches also exhibited specific patterns of daily movements, spending most of the day at the foraging site before returning to the roosting area (Lewis 2007), as do the BTF at Site 2. Radio-tracked Gouldian finches moved an average of 3 km/day between foraging areas and water sources (Woinarski and

Tidemann 1992, Tidemann 1993) while the greatest distance a BTF moved from its roosting site was 1.5 km. This difference may be related to the methodology used to estimate movements or it may be a function of environment. Inter-site differences in local movement patterns have also been found in the Savannah Sparrow (*Passerculus sandwichens*) in a grassland vegetation in south-eastern Arizona, with sedentary behaviour at one site and high mobility at another (Gordon 2000). The locations of the foraging and roosting sites influence home range sizes for Savannah sparrows, with roosting sites lying outside of foraging areas as birds have different requirements for each activity (Ginter and Desmond 2005). Some BTF show similar behaviour. Fine scale studies are required in foraging and roosting areas to determine their ecological requirements. Over the time frame for which BTF were radio-tracked they used small home ranges and this can indicate reliability of resources (Ginter and Desmond 2005).

Activity and time of the day

BTF spent time early in the morning in nest maintenance (carrying nest material, repairing existing roosting nests) while foraging activities dominated later in the morning after nesting maintenance activities were completed. In the afternoon, during the hottest part of the day, birds rested in shade, usually as a single flock. In comparison, crimson finches formed flocks of 13.2 ± 0.5 individuals during the non-breading season but formed smaller groups (4.2 ± 0.1) or pairs during breeding season (Milenkaya et al. 2011). In the early 1990s, a BTF flock of about 150 individuals was recorded on the Townsville Coastal Plain (Mitchell 1996). In comparison, no large flocks were observed while this study was in progress, almost 20 years later, nor reported to me or to the Black-throated Finch Recovery Team (Vanderduys pers. comm.). This difference in maximum flock size is likely due to a declining population. Considering the time frame of our study and activities during the day, BTF flocks were small early in the morning or when in the roosting area but birds congregated to form a larger flock in the foraging area (Site 2). Similar behaviour was observed at Site 1 where small groups would roost separately but these would aggregate during the day. Mitchell (1996) also observed flock size vary throughout the day, with larger flocks seen between 100 and 300 minutes after sunrise. He observed that BTF would forage in the morning (though the flocks were smaller during this study) (Mitchell 1996). Although some small differences in the pattern might have happened due to different estimators used to calculate the home ranges, the differences in dynamics of flocks over the course of a day are consistent with radio-tracking data.

Habitat selection

Determining the resources and habitats that are preferentially used by an animal population is important for understanding how animals meet their requirements (Manly et al. 2002). In this study the vegetation communities used preferentially by BTF differed between sites. Six birds concurrently radio-tracked in the late dry season of 2014 preferentially used *Eucalyptus crebra* woodland (RE 11.3.30). At Site 1 the vegetation preferentially used by the birds was *Eucalyptus platyphylla* woodland (RE 11.3.35) and they apparently avoided *Eucalyptus crebra* woodland, while at Site 2 they apparently avoided *Eucalyptus platyphylla* woodland. Manly et al. (2002) pointed out that if habitats that are less favoured are the only ones available then species may comprise a large proportion of those used.

Resource and habitat selection is often affected by season, sex, age, animal behaviour and daily activity pattern. The differing use of vegetation communities across sites could be due to non-uniform resource availability in the landscape (seeds patchily distributed in the landscape) so the use of those resources by animals will change with their availability. Alternatively the selection of habitat at the RE level on both sites could also be related to the condition of the RE including grazing intensity or presence of invasive plants among others. Additionally, the REs used to describe habitat for the BTF may not be sufficiently discerning. Fine scale patterns in the landscape might determine habitat choices. For instance, in both areas I observed birds using patches of *Melaleuca* spp. to rest during the hottest part of the day. They avoided patches with the introduced shrub *Stylosanthes scabra* and forage preferentially in grassy areas with patches of bare ground; a burnt patch at Site 1 was intensively used for foraging by BTF and co-occurring granivorous birds. However these features are not captured in descriptions of REs.

Observations from this study showed that BTF were spending most of their time during the day foraging or resting in *Eucalyptus crebra* woodland (RE 11.3.30) at Site 2, and going to other areas, such as *Eucalyptus platyphylla* woodland (RE11.3.35), to roost. They foraged in both vegetation communities. Vegetation structure and composition might have influenced their choices. There is some grazing activity at Site 2 within the patch of *Eucalyptus crebra* woodland that keeps the vegetation less dense while in the roosting areas the vegetation density is higher. The only published management guide for BTF habitat indicates that livestock grazing can be compatible with persistence of BTF; the fact that the foraging area of the largest flock of BTF recorded in this study was in a grazed area corroborates this. However, further studies must focus on the intensity of grazing that is ideal for BTF (i.e. what grazing regime produces vegetation that delivers both a good supply of seed and a foraging environment in which they can access it). Field observations showed that BTF used mango plantations mainly to rest during the hottest part of the day and BTF were observed in this habitat in only one period (early dry season) of tracking, and not observed again using this vegetation type.

Conclusion

Knowledge of movement patterns and habitat selection by a species is a prerequisite for understanding their ecological needs and thus planning realistic conservation strategies (Brandt and Cresswell 2008). This is the first study of movement patterns, home range sizes and habitat selection of BTF and it provides important basic information about the species' ecology. As for Gouldian finches (Lewis 2007), to thoroughly understand long-distance, local and daily movement patterns of BTF, it is essential to have a thorough knowledge of which resources are required throughout the seasonal cycle as well as availability and distribution of those resources. This study showed BTF had fixed home ranges over short time scales with at least a few individuals moving comparatively long distances; overall birds were resighted near the water sources where they were mist netted and banded. There are some features in the landscape that might influence habitat selection by BTF but scales finer than the RE should be investigated. BTF in our study are mostly residents and this information should be acknowledged in conservation and management actions.

Radio-tracking surveys also point to the issue of false absences of BTF. Detectability of a species is correctly identifying the presence of one species in a particular area (Thompson 2002). BTF seems to have a low detectability as even birds with radio transmitters were sometimes not sighted or heard in the area. Future studies should be carried out to quantify detectability under different circumstances to more reliably ascertain when the species is absent.

Major areas of the state of Queensland are undergoing significant landscape changes related to residential development, rural intensification and mining. The impact of those changes on BTF has been little studied and little is known about the relative importance of different vegetation types (habitat and micro habitat features) for the birds. In this study, there were significant differences between the two sites and 15 birds monitored, differences in home range sizes, daily movement and habitat selection. Given that there is so much variation in space and time, it is imperative to develop a much better understanding of resource use under different environmental and climatic conditions. Local and regional conservation plans need to address BTF needs at the large scale (whole of range) and in relation to the vegetation features that are vital to the birds.

Chapter 3

Habitat requirements of an endangered granivorous bird in tropical savannas, the Black-throated finch *Poephila cincta cincta*[2]

Abstract

Vegetation structure and composition influence the way animals use the habitat, and knowledge of key environmental requirements of those animals is paramount for the development of conservation and management actions. This study aimed to understand the requirements of the Black-throated finch southern subspecies (BTF), an endemic granivorous bird of north-eastern Australia. Bird and vegetation surveys were undertaken at 10 sites south of Townsville, Australia. Conditional inference trees were used to verify the main features associated with BTF flocks, and then specific hypotheses were tested. BTF mean flock size was 4.1 birds, with a maximum observed being 40 birds. BTF flocks were most closely associated with (1) higher cover of native grasses, (2) lower cover of shrubs with the presence of dead trees or (3) less cover of shrubs but vegetation density higher in the lowest stratum (0-20 cm height) and (4) high cover of *Eragrostis* spp. Small flocks were associated with: (1) low percentage of native grasses and low percentage of facultative perennial plants, (2) high shrub cover and high total arboreal richness, (3) high shrub cover and low grass richness, (3) low percentage of *Eragrostis* spp. and *Setaria surgens*, and (4) low percentage of *Eragrostis* spp. and low grass richness. Models partially supported five hypotheses and overall it was observed BTF flocks had a significantly positive relationship with *Aristida warburgii*, *Eragrostis* spp, bare ground, visibility, high cover of native and non-native species and high abundance of dead trees. Flocks were negatively associated with shrub abundance, shrub cover, large trees abundance and total vegetation cover (ground cover). BTF require different aspects of ground cover and woody plant cover. They prefer areas with low vegetation density, and low shrub cover and abundance, but the presence of a medium strata (e.g. dead trees or *Acacia holocericea*) may benefit the birds by assisting them to access foraging sites.

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[2] This chapter will be submitted as a paper:
Rechetelo, J., Wilcox, C., Moloney, J., Reside, A.E., Hardesty, B.D. and Grice. A.C. 2016. Habitat requirements of an endangered granivorous bird in tropical savannas, the Black-throated finch *Poephila cincta cinct*a.

Introduction

Identifying the habitat requirements of a species, and understanding how animals use habitat, are prerequisites for successful conservation outcomes (Macnally 1990, Yen et al. 2011). Vegetation structure and composition are important habitat elements driving the species composition of most terrestrial faunal communities (Wiens and Rotenberry 1981, Cody 1985, Hewson et al. 2011). It also influences the way animals use the habitat as they require specific resources for different aspects of their life histories (Block and Brennan 1993). While vegetation structure provides information about the availability of resting, roosting, nesting and foraging areas, plant composition is often associated with the type and timing of resource availability (Augenfeld et al. 2008, Deppe and Rotenberry 2008). The influence of habitat features – vegetation structure and composition – on bird assemblages has therefore been an important and ongoing theme in ecology (Rotenberry 1985, Martin 1992, Block and Brennan 1993, Jayapal et al. 2009, Fuller 2012b).

Studies of how vegetation structure and composition affect bird communities in savanna landscapes have been conducted in a variety of locations worldwide (Dean et al. 1999, Skowno and Bond 2003, Tassicker et al. 2006). Although Australian savannas are relatively intact (Franklin 1999, Woinarski et al. 2007), contemporary land uses are impacting biodiversity (Werner 1991, Woinarski 2000) and major faunal declines have been associated with changes in land use. Therefore, many studies focus on how specific aspects related to land use - woody thickening, changes in fire regimes, grazing/pastoralism, introduced species (plants and animals) or a mix of different land uses – are affecting the fauna (Woinarski 1990, Woinarski and Recher 1997, Woinarski et al. 1999, Woinarski et al. 2004, Andersen et al. 2005, Whitehead et al. 2005, Kutt and Woinarski 2007, Kutt and Martin 2010, Cook and Grice 2013, Woinarski and Legge 2013).

Land use changes negatively affect various fauna guilds (Tassicker et al. 2006), but granivorous birds feature prominently in species declines (Franklin 1999, Franklin et al. 2005, Woinarski et al. 2005). Given the sharp rate of decline among many granivorous birds, it is important to understand their ecological requirements and how the structure and composition of the landscape influence them, particularly in the case of endangered species, where this information can help with conservation and management actions.

The Black-throated finch, *Poephila cincta*, is an endemic granivorous Estrildid finch of north eastern Australia and the southern subspecies (*P. cincta cincta*, herein referred to as BTF) has undergone an approximately 80% range contraction (BTFRT 2007a, Buosi 2011). BTF occur in grassy open woodlands in a broad matrix of tropical savannah habitats; recent studies showed that about 60% of the remaining suitable habitat for BTF may be at risk from mining developments (Vanderduys et al. 2016). Habitat management is therefore crucial to its conservation and recovery (Buosi 2011). However, little is known about their specific habitat needs and the conservation of the remaining populations requires a sound understanding of these needs in the face of land use intensification in northern Australia. The aim of this study was to identify the environmental factors, particularly relating to vegetation structure and composition that are important in determining BTF occurrence.

Methods

Conservation Status

The BTF is listed as endangered under the Federal *Environment Protection and Biodiversity Conservation Act* (EPBC 1999), the Queensland *Nature Conservation Act* (Queensland Government 2006a) and the New South Wales *Threatened Species Conservation Act* (New South Wales Government 1995b). Significant populations of BTF are known to remain in two main areas, the Townsville coastal plain (northeast Queensland) and

Galilee basin (inland central Queensland) (Vanderduys et al. 2016). BTF is identified as a High Priority species under the Back on Track species prioritisation framework approach by the Department of Environment and Heritage Protection (Buosi 2011, Department of Environment and Heritage Protection 2015).

The study area

This study was conducted from 2011-2014 in the vicinity of Lake Ross, south of Townsville, Queensland, north eastern Australia (Fig. 3.1). The study area is between 19º24'26.29" S and 19º34'21.33" S and 146º45'09.46" to 146º50'11.38"E. This area is within the Townsville Coastal Plain sub region of the Brigalow Belt Bioregion (Sattler and Williams 1999). *Eucalyptus* open-woodlands are the dominant vegetation type, but areas of *Melaleuca* low open-woodland and open forest are also common (Murtha 1975, Sattler and Williams 1999). The ground vegetation layer is naturally dominated by native kangaroo grass (*Themeda triandra*), black speargrass (*Heteropogon contortus*) and northern canegrass (*Mnesithea rottboellioides*) (Murtha 1975, The State of Queensland 2014). Introduced grasses, shrubs and trees are common in many cleared and uncleared areas. For example, Indian jujube (*Ziziphus mauritiana*) dominates the shrub or tree layer, while stylos (*Stylosanthes* spp.), snakeweed (*Stachytarpheta jamaicensis*), Guinea grass (*Megathyrsus maximus*), Sabi grass (*Urochloa mosambicensis*) or grader grass (*Themeda quadrivalvis*) (Smith 2002, Grice et al. 2013) dominate the ground layer in many areas.

Rainfall of the study area is markedly seasonal with average annual rainfall of 1,157 mm, most of which falls during the six months between November and April (Murtha 1975, Bureau of Meteorology 2015). The study site is located within the Brigalow Belt North Bioregion, a biodiversity hotspot containing important habitat for rare and threatened species (Environmental Protection Agency 2008).

Sites

Ten study sites in the vicinity of Lake Ross, an artificial water storage south of Townsville, were selected (Fig. 3.1). Sites are 10 - 20 km from the urban edge of Townsville, which is not only Australia's largest tropical city, but is rapidly expanding. This study area was chosen because it supports a substantial and readily accessible population of BTF (BTFRT/NRA 2011). Site locations were selected based on historical records of BTF (e.g. Black-throated finch Recovery Team database), local knowledge, and personal reconnaissance surveys. Final site selection was also based on physical accessibility, spatial separation (different areas around Lake Ross) and landholder permission.

All sites were surveyed during the dry season (the 2013 dry season lasted longer than normal; rain started in January 2014). Five sites were on restricted access public land in the Lake Ross catchment area and five were on private properties. The Lake Ross sites had a low intensity of human activity though there was some historical and continuing prescribed burning and livestock grazing (Appendix B; Table B.1). Private property sites were continuously or intermittently grazed by cattle or subject to uses other than livestock grazing or agriculture. Under the Australian vegetation classification system (The State of Queensland 2014), four sites were mapped as non-remnant and the remaining sites mapped within three Regional Ecosystems (herein referred as RE; Table 3.1). Three sites were within *Eucalyptus platyphylla-Corymbia clarksoniana* woodland RE, two sites in the *Melaleuca viridiflora* with occasional *M. argentea* and *M. dealbata* woodland RE and one site was in Riverine wetland or fringing riverine wetland RE (Table 3.1). All REs in the study area are classified as Least Concern under the Vegetation Management Act 1999 (The Queensland Government 1999). BTF were sighted and known to use eight of the sites (based on local information and the author's observations of water sources frequently used by the birds) whereas birds were never sighted at the other two sites.

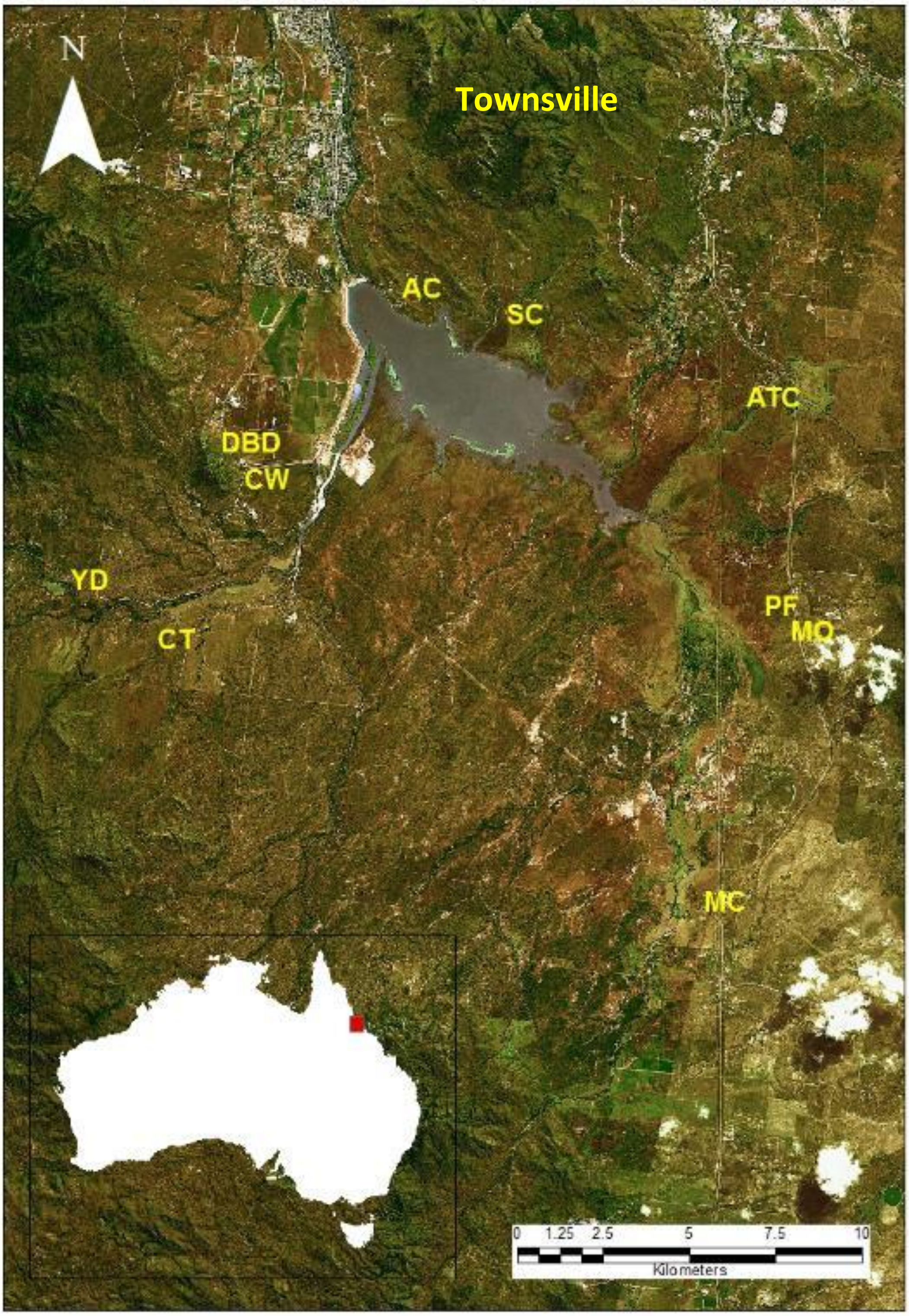

Fig. 3.1. Study area around Lake Ross, southeast of Townsville, eastern Australia, with 10 study sites (Codes: YD, CT, DBD, CW, AC, SC, ATC, PF, MO and MC).

Bird sampling

Bird surveys were carried out between November 2011 and January 2014 (27 months), during both wet and dry seasons. Sites were visited weekly or monthly although, during the wet seasons, some sites were inaccessible. Surveys involved 30 min to 2 hours at each site as well as watching creeks and other water sources where BTF would come to drink. Absolute counts of BTF were undertaken along transects; all BTF seen or heard were recorded. Presence or absence during each visit constitute a record.

Table 3.1. Regional Ecosystem code, estimated extent with Management Act class, biodiversity status and description of Regional Ecosystem present in the study sites surveyed in BTF habitat areas, south of Townsville (The State of Queensland 2014).

Regional Ecosystem	Estimated extent	Biodiversity status	Description
11.3.12	In 2011, remnant extent was > 10,000 ha and >30% of the pre-clearing area remained	No concern	*Melaleuca viridiflora* with occasional *M. argentea* and *M. dealbata* woodland. Widely cleared for cropping and pasture
11.3.25b		Of concern	Riverine wetland or fringing riverine wetland. *Melaleuca leucadendra* and/or *M. fluviatilis*, *Nauclea orientalis* open forest. Weeds such as rubber vine (*Cryptostegia grandiflorus*) and lantana (*Lantana camara*) have invaded several areas
11.3.35		No concern	*Eucalyptus platyphylla*, *Corymbia clarksoniana* woodland. Invaded by *Ziziphus mauritiana* (chinee apple) and *Cryptostegia grandiflora* (rubber vine) in several districts
Non Remnant		Areas similar to the RE close by but cleared for land use purposes or areas that had some disturbance but present re-growth of local native species and some invasive	

Vegetation survey

Vegetation surveys were conducted using a modified BioCondition methodology (Fig. 3.2) (Eyre et al. 2011, Kelly et al. 2011), incorporating additional measures likely to be relevant to granivorous birds (Robel et al. 1970). The BioCondition method involves 100 m line transects along which 5 x 1 m^2 quadrats are placed at 0 m, 25 m, 50 m, 75 m and 100 m where ground cover information (grass and forb structure and composition) is recorded (Table 3.2). Additionally, woody plant cover information – trees, shrubs, vines – was recorded along 10 m wide belt transects (5 m each side of the main line transect). Five transects were located at each site, totalling 50 transects (250 quadrats). The minimum distance between transects was 50 m. Grasses, forbs, shrubs, vines and trees were identified to species or genus level. Trees were classified into three size categories by diameter at breast height (DBH); <10 cm (small trees), 10-40 cm (medium trees) and >40 cm (large trees) (Table 3.3). Vegetation surveys were conducted between July and December 2013 (three in July, three in August, two in November and two early December). Areas with no bird records have historical relevance (Appendix B, Table B.1).

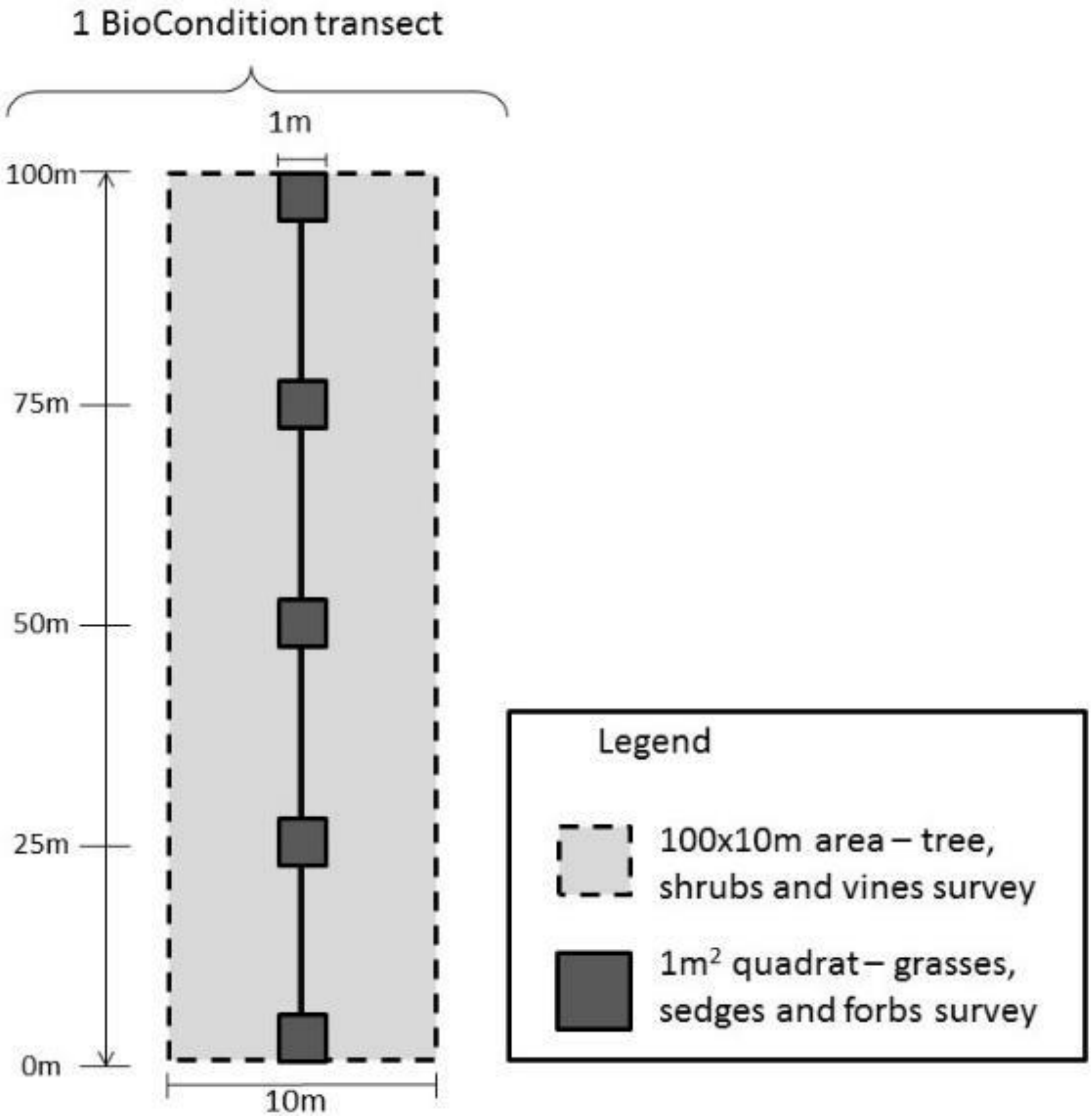

Fig. 3.2. BioCondition methodology: 100 m transect, 5 x 1 m2 quadrats (dark gray) placed along the transect to assess ground cover (grasses and forbs, bare ground, rock, litter) and a wider area (light gray) 10 x 100 m to assess Woody Plant Cover (trees, shrubs, dead trees standing, dead trees on the ground and fallen wooden material). Visual obstruction and vegetation density were measured using a Robel pole (1 m height) and placed in the centre of each quadrat. Five transects were mapped at each study site.

Table 3.2. Ground cover environmental variables measured in BTF habitat in 10 areas around Lake Ross. Ground cover data were recorded in $5m^2$ quadrats along a 100 m transect. A total of 20 ground cover structure variables were recorded.

Variables			Definition
Ground Cover Structure (GS)	GCD		Ground cover (vegetation) density: *n* leaves/stems intercepted at a vertical point recorded in 20cm height increments from ground level. Six classes were analysed based on the stratum of the vegetation: GCD 0-20 cm, GCD 20-40, GCD 40-60, GCD 60-80, GCD 80-100 and GCD >100 cm. Unit: counts of 'hits'. Mean was calculated for each class.
	VO		Visual obstruction – a 1m pole was placed in the centre of the quadrat and an observer estimated the visibility from an eye height of 1.0m and 4m from the pole (e.g. in an area with no ground cover, 100% of the pole was observed and this was the visibility recorded). VO was estimated 3 times and the mean was calculated.
	Total 100%	BG	% bare ground in a $1m^2$ quadrat (BG, rock, litter and VC added 100%) estimated in all 5 quadrats in the transect
		Rock	% rock (as above)
		Litter	% of litter (as above)
		VC	% of vegetation cover – grasses, forbs, sedges (as above)
	G_NSpp		Total *n* species (grass/forb/sedges) within that quadrat
	GrassNoSp		*n* of grass species only
	Life form		% of grasses and % of forbs (2 classes)
	Status		% of native (PerNative) and non-native (PerExotic) species (2 classes)
	Life cycle		% of annual, % perennial and % facultative perennial plants (3 classes)
Ground cover composition (GC)			Grasses, forbs and sedges identified to species where possible, otherwise genus, or family

Table 3.3. Woody plant cover environmental variables measured in BTF habitat. Woody plant cover data were recorded in a 10x100m area.

Variables		Definition
Woody Plant Cover Structure (TS)	T1 height	Average of the highest trees (T1) in the area at canopy level (m)
	T2 height	Average height tallest sub-canopy layer (T2) (m)
	T3 height	Average height lowest sub-canopy layer, taller than 2m (T3) (m)
	SH height	Average height of shrubs in the area (m)
	T1 cover	Foliage projective cover T1 trees, interpreted as a vertical projection straight on the ground (m)
	T2 cover	Foliage projective cover T2 trees (m)
	T3 cover	Foliage projective cover T3 trees (m)
	SH cover	Foliage projective cover of shrubs (m)
	Shrub abundance	*n* individual shrubs in a 100 x 10 m^2 area (5m from each side of the transect)
	Shrub richness	*n* shrub species (as above)
	Small trees abundance	*n* individual trees DBH < 10 cm
	Small trees richness	*n* tree species DBH < 10 cm
	Medium trees abundance	*n* individual trees DBH 10-40 cm
	Medium trees richness	*n* tree species DBH 10-40 cm
	Large trees abundance	*n* individual trees DBH > 40 cm
	Large trees richness	*n* tree species DBH > 40 cm
	Total arboreal abundance	Total *n* tree and shrubs individuals
	Total arboreal richness	Total *n* tree and shrubs species
	Basal area	Basal area measured in the centre of each transect (50m mark) by using cruise master prism
	DTS	*n* dead trees that were still standing and used as perches (e.g. burnt trees with no leaves or fruits)
	DTG	*n* dead trees that were on the ground (e.g. trees that have fallen after a fire or a storm)
	FWM	Fallen woody material – *n* of logs larger than 10cm diameter and 0.5 m long in an area 50 x 10 m along the tape measure
Woody Plant Cover Composition (TC)		Trees, shrubs and vines were identified to species where possible, though some could only be assigned to genera

Data Analysis and Bird habitat models

I compared the differences in habitat variables across sites using ANOVA. Shapiro Wilk Normality test and Bartlett's test for Homogeneity of Variances were used to test ANOVA assumptions. Tukey HSD was used for post hoc tests. If assumptions were not met, I used Kruskal-Wallis tests. BTF flock size was related to environmental variables first using Conditional Inference Trees (Hothorn et al. 2006, Hothorn et al. 2014a) and then specific hypotheses were tested using a negative binomial generalized linear model (McCullagh 1984, McCulloch 2000).

Conditional Inference Trees estimate a regression relationship by binary recursive partitioning in a continuous, censored, ordered, nominal and multivariate response variables framework. Conditional inference trees are not affected by over-fitting and are unbiased with regard to the types of explanatory variables used (Hothorn et al. 2006, Strobl et al. 2007, Johnstone et al. 2014). They use a statistically-determined stopping criterion to determine where splitting is no longer valid (an a priori P value). Because of the a priori P, it is not necessary to 'prune' the trees (Johnstone et al. 2014). The P value denotes the deviation from the partial null hypothesis, and is produced by assigning weights to nodes (Hothorn et al. , Hothorn et al. 2010, 2014a, Hothorn et al. 2014b, Johnstone et al. 2014). The data were recursively split to produce an inverted tree (Hothorn et al. 2006, Hothorn et al. 2014a, Hothorn et al. 2014b, Johnstone et al. 2014), with the first node being the tree's root. All possible binary splits of the response variable (flock size) were assessed for each potential explanatory variable. In this study, explanatory variables were grouped as (1) ground cover structure variables; (2) woody plant cover structure variables; (3) ground cover and woody plant cover structural variables; (4) grass and tree composition; and (5) all variables for grass and tree structure and composition. The variables were grouped to better investigate how different aspects of vegetation structure and composition were affecting the birds.

Negative binomial generalized linear models was used to test specific hypotheses (Table 3.4). BTF habitat hypothesis were based on current ecological knowledge of BTF (BTFRT 2007a, Buosi 2011). In addition, the square of bare ground cover was considered in three of our models in order to account for curvilinear relationships and to seek the optimal habitat profile. For example, if bare ground percentage was too high, there may be too few grasses to provide food resources, whereas if ground cover is too high, the bird's access to those resources would be impaired. The bird data were not normally distributed so I assumed a negative binomial distribution and size of flock was related to explanatory variables. Model ranking was based on the AICc values (Burnham and Anderson 2002). I evaluated the goodness-of-fit and overdispersion of each model. Where models had substantial support (ΔAICc < 2) (Burnham and Anderson 2002), parameter estimates and standard errors of alternative models were examined to assess whether inference made from those models would be different. Absence data were used in the models as predictions are more accurate (Brotons et al. 2004). All analyses were performed with the statistical package R (R Core Team 2014) and the libraries 'Party' (Hothorn et al. 2014b), 'lattice' (Sarkar 2015), 'MASS' (Ripley et al. 2014), 'MuMIn' (Barton and Barton 2014) and 'qpcR' (Ritz and Spiess 2008).

Table 3.4. Hypotheses considering current knowledge of aspects of the ecology of BTF. Groups of variables: type of variables used in the hypotheses. GS: ground cover structural variables. GC: ground cover compositional variables. TS: woody plant structural variables. TC: wood plant cover compositional variables.

Hypothesis: BTF flocks will be larger in areas with:	Group of variables
bare ground, more grass cover (%) + where there is overall more grass species (high richness) and less forb cover (%)	GS
higher vegetation cover (%), more native grass species (%), high visibility and less cover of exotic grass species	
higher cover of grass species of *Chrysopogon fallax* + Chloris + Eragrostis + *Aristida warburgii* are more abundant	GC
the percentage of *Chrysopogon fallax* and Eragrostis spp. more abundant, where there is more grass species (high richness) and more bare ground	GC + GS
high abundance of: dead trees standing, medium trees and larger trees and lower abundance of shrubs	TS
high abundance of dead trees standing but lower shrub cover and total arboreal abundance	
high abundance of *Eucalyptus platyphylla* and *Melaleuca nervosa* but low abundance of *Ziziphus mauritiana*	TC
more bare ground, more grass cover (%) and grass species (high richness), high large trees abundance and high abundance of *Eucalyptus platyphylla*	GS + TS + TC

Results

Black-throated finch flocks

Three hundred and forty records – BTF presence (flock size) or absence – were obtained between November 2011 and January 2014. Mean flock size across all records was 4.1 birds (Fig. 3.3; Table 3.5). Flock size was significantly different between sites ($F_{9,330} = 8.7654$; $P < 0.001$).

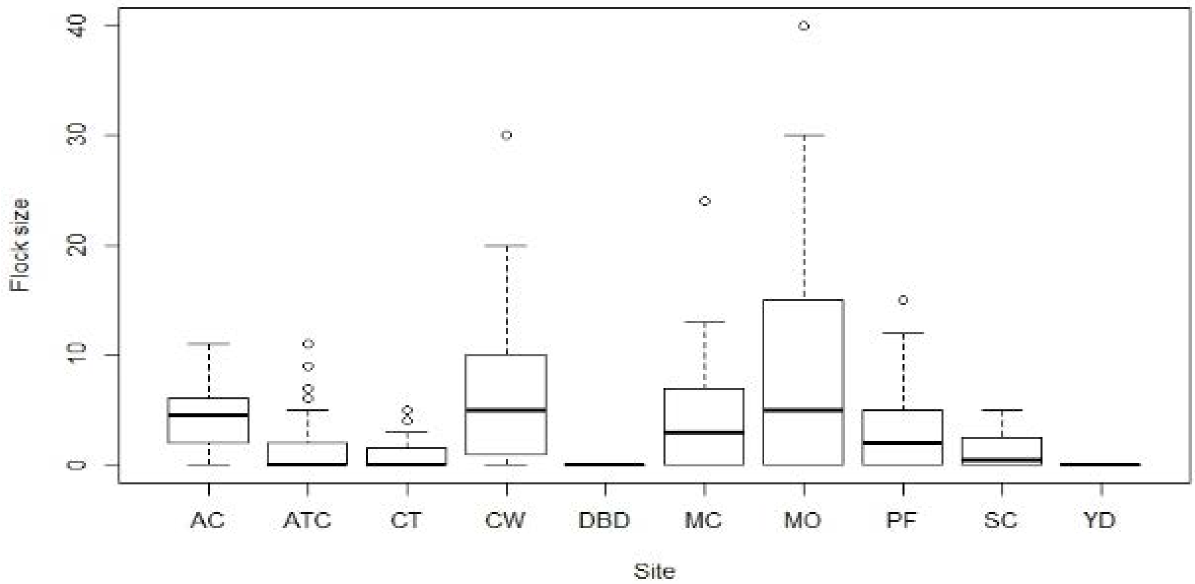

Fig. 3.3. BTF flock size at 10 different sites located in the vicinity of Lake Ross.

Table 3.5. Sites, number of visits per site, number of months each site was surveyed, mean, maximum, minimum, median and standard deviation of BTF flock size at 10 different sites in the vicinity of Lake Ross.

Sites	***N* visits**	***N* months**	**Flock mean**	**Flock max**	**Flock min**	**Flock median**	**Flock SD**
AC*	20	7	4.5	11	0	4.5	3.22
ATC	56	20	1.5	11	0	0	2.68
CT[1]	8	10	1.0	5	0	0	1.86
CW	65	16	5.9	30	0	5	5.92
DBD[1]	16	7	0	0	0	0	0
MC[1]	50	17	4.2	24	0	3	4.82
MO[1]	48	13	9.3	40	0	5	11.66
PF	53	19	3.0	15	0	2	3.43
SC*	8	5	1.4	5	0	0.5	1.84
YD[1]	12	9	0	0	0	0	0

* Inaccessible during wet season

[1] Private properties – accessibility subject to landholder permission

Summary of habitat characteristics

Mean visibility was 70% (min = 20% and max = 100%), with significant differences between sites ($F_{9,40} = 11.57$; $P < 0.001$). Basal area ($F_{9,40} = 10.03$; $P < 0.001$), and abundance of standing dead trees ($F_{9,40} = 9.61$; $P < 0.001$), fallen dead trees ($F_{9,40} = 2.74$; $P < 0.01$), FWM ($F_{9,40} = 11.19$; $P < 0.001$) and ground cover also differed between sites (Rock: $F_{9,40} = 2.6$; $P < 0.05$; Bare Ground: $\chi^2 = 23.93$, df = 9, $P = 0.004$; Litter: $\chi^2 = 26.68$, df = 9, $P = 0.001$; Vegetation cover: $\chi^2 = 24.1$, df = 9, $P = 0.004$).

A total of 58 grasses/forbs were recorded at the study sites: 26 grasses (five annual, 15 perennial and six facultative perennial; 13 native to Australia, nine non-native and four unknown) and 32 forb/sedge species (nine species being annual, 12 perennial and 10 facultative perennial; 16 were native to Australia, 11 non-native and four unknown) and one unknown (Appendix B, Table B.2). The mean number of grass/forb species recorded per transect was 6.1 (min = 1, max = 14; $F_{9,40} = 3.599$; $P < 0.01$).

Forty-two species of trees/shrubs were identified at the study sites (Appendix B, Table B.3); 31 tree species in total and 11 shrubs/vines. Of the tree species, 28 were native to Australia, one was introduced and two were unknown. Of the shrubs and vines, eight were native to Australia and three introduced. The tree/shrub richness per transect varied from one to 14 (mean = 5.7) and differed significantly between sites ($F_{9,40} = 6.44$; $P < 0.001$). Tree and shrub richness per site varied from 2 to 23 species. A total of 4,968 individual trees/shrubs were recorded (Appendix B, Table B.3). The tree/shrub abundance per site varied from 37 to 1,101 (Appendix B, Table B.4). Tree/shrub abundance per transect ranged from one to 372 individuals (mean = 99.4) and differed significantly between sites ($F_{9,40} = 6.88$; $P < 0.001$). Ten species represented 89.2% of total abundance, including seven trees (*Corymbia tessellaris*, *C. dallachiana*, *Eucalyptus platyphylla*, *Lophostemon grandiflorus*, *Melaleuca viridiflora*, *Planchonia careya* and *Petalostigma pubescens*) and three shrubs (*Acacia leptostachya*, *Lantana camara* and *Ziziphus mauritiana*). Of the woody plants, species classified as shrubs or vines represented 17% of total arboreal abundance. Among shrubs and vines, only three species represented 85% of total shrub/vine abundance: *Z. mauritiana* (42.9%), *A. leptostachya* (28.7%) and *L. camara* (13.9%). Of the woody plants, 48% were small trees, 39.9% were shrubs, 11.1% medium trees and 0.9% large trees.

Flock-size associations with environmental factors

Ground cover structural variables: Larger BTF flocks were associated with areas with more than 13.5% of native grass species (Node 5); and where the percentage of native grasses was less than 13.5% but the percentage of facultative perennial grasses was higher than 2.6% (Node 4). BTF flocks were smaller in areas with less cover of both native and facultative perennial grasses (Node 3) (Fig. 3.4).

Woody plant cover structural variables: larger BTF flocks were associated with foliage projective cover of shrubs less than 0.7 m and an abundance of dead trees larger than 0.2 (Node 4). Medium size flocks were associated with less shrub cover and lower abundance of dead trees (Node 3) or more shrub cover and lower total arboreal richness (Node 6). BTF flocks were smallest in areas with greater shrub cover and high total arboreal richness (Node 7) (Fig. 3.5). A similar conditional inference tree is generated when woody plant cover structural and compositional variables were evaluated together; the only difference is in the second node - DTstanding is replaced by the species *Acacia holosericea* (Appendix B, Fig. B.1).

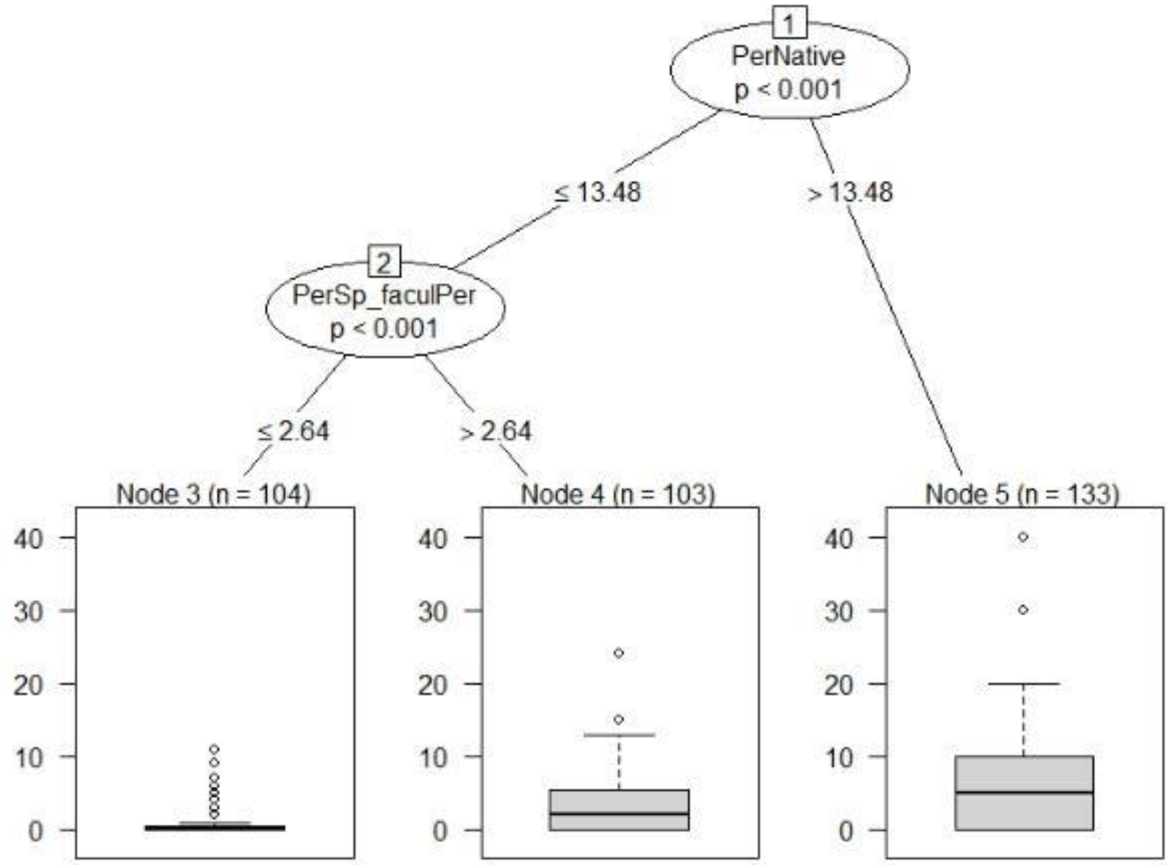

Fig. 3.4. Conditional inference tree examining BTF flock size at different sites using ground cover structure as explanatory variables (number of variables = 20). Flock size was best classified into three groups (leaves) considering this subset of data. Node 3 mainly comprises sites with small flock sizes, whereas Nodes 4 and 6 comprise sites with larger BTF flocks. Nodes: variables that best explained flock size differences. Node 1: percentage of native plants. Node 2: percentage of facultative perennial plants. Boxplots show median, ranges and upper and lower quartiles for populations in which no further splitting was possible. e.g. flock size split by percentage of native species; sites with high percentage of native plants had a mean flock size greater than sites with a low percentage of native plants. The vertical axis represents bird abundance/flock size.

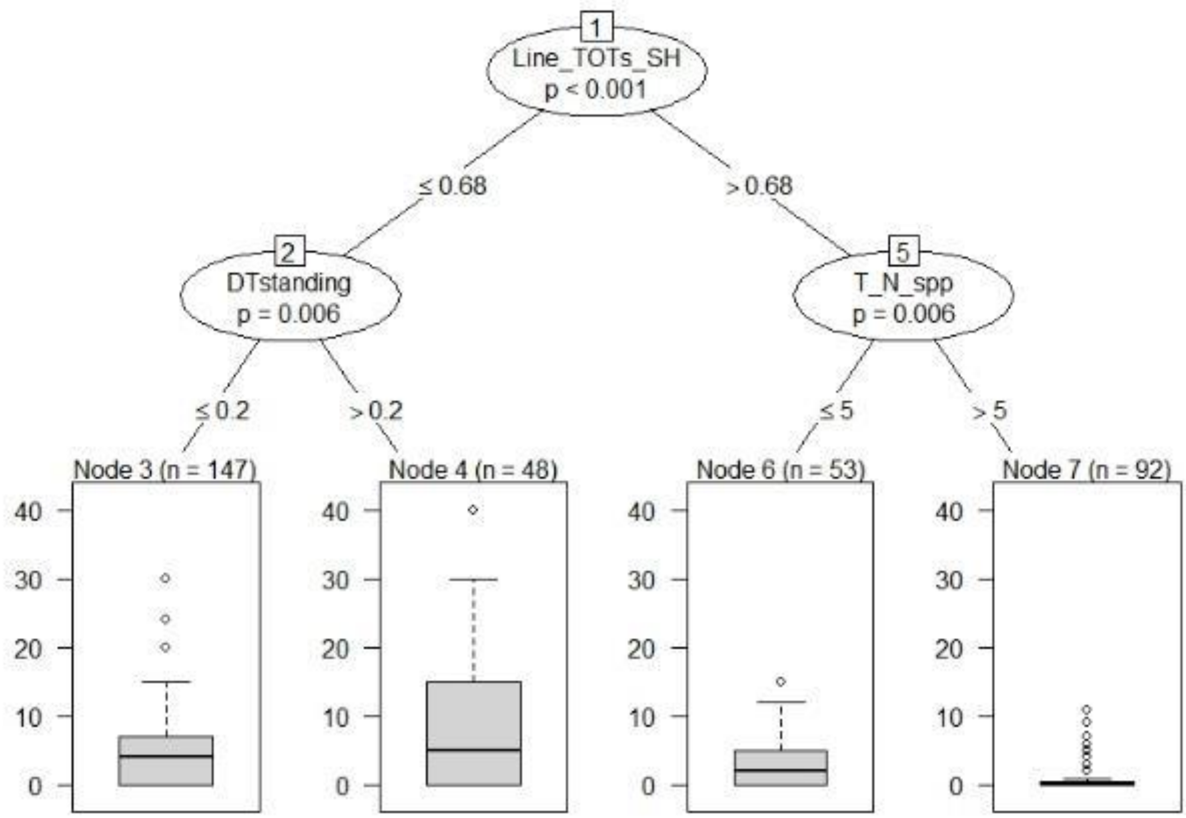

Fig. 3.5. Conditional inference tree examining BTF flock size in different sites using woody plant cover structure as explanatory variables (number of variables = 22). Flock size was best classified into four groups (leaves) considering this subset of data. Node 7 is primarily sites with small flock sizes whereas Nodes 3 and 6 were associated with medium flock sizes. Node 4 is associated with the largest BTF flock. Node 1: foliage projective cover of shrubs. Node 2: number of dead trees that were still standing and used as perch. Node 5: total arboreal richness.

Ground cover and woody plant cover structure: larger BTF flocks still remain associated with foliage projective cover of shrubs less than 0.7 m and vegetation density higher at first class (Node 4). Median size flocks were associated with lower shrub cover and vegetation density at the first class (Node 3) or higher shrub cover with high grass species richness (Node 7). Flocks were smallest when shrub cover was greater and grass species richness lower (Node 6) (Fig. 3.6).

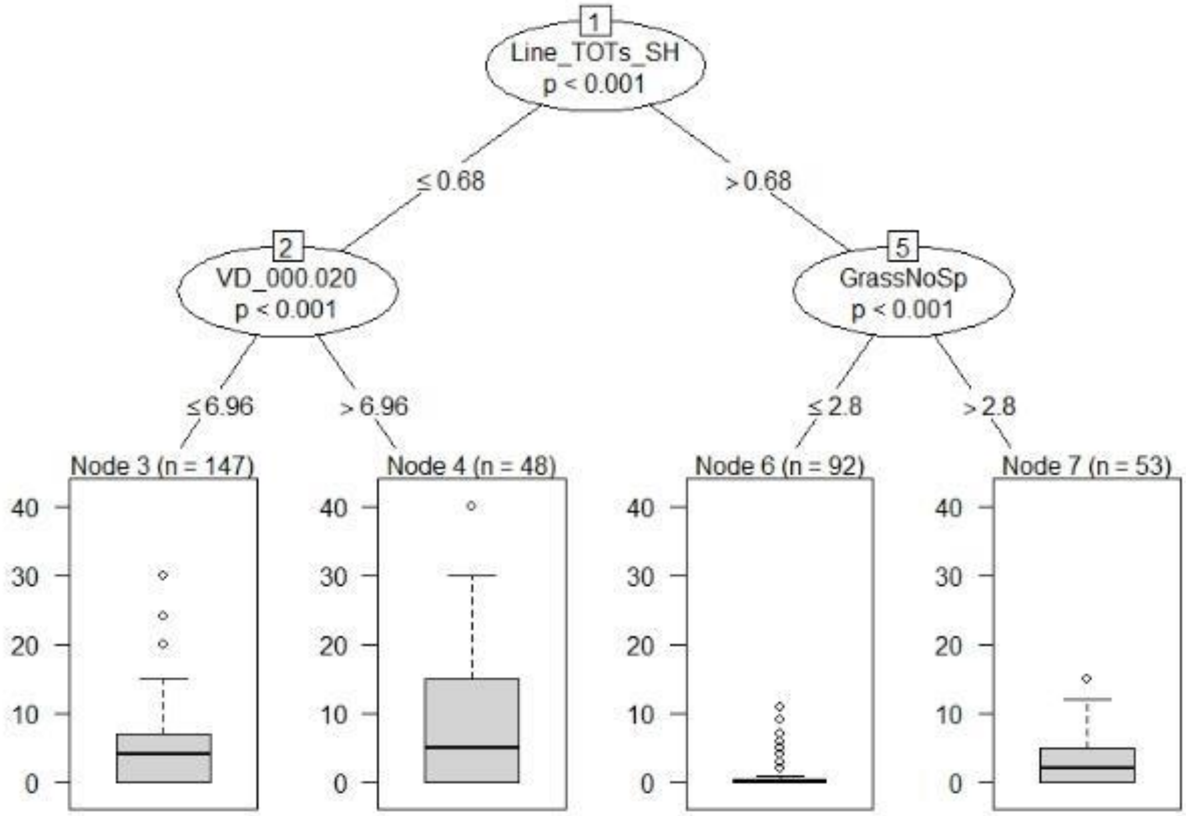

Fig. 3.6. Conditional inference tree examining BTF flock size in different sites using both ground cover and woody plant cover structure as explanatory variables (number of variables = 42). Flock size was best classified into four groups (leaves) considering this subset of data. Node 6 is small flocks, nodes 3 and 7 medium flocks and node 4 larger flocks. Node 1: foliage projective cover of shrubs. Node 2: number of leaves/stems intercepted at a vertical point recorded in the first class of vegetation density (0- 20 cm). Node 5: number of grass species.

Ground cover and woody plant cover composition (excluding forbs): larger BTF flocks were associated with high percentages of *Eragrostis* spp. cover (Node 5) and when the percentage of *Eragrostis* spp is smaller than 1.2%, flocks will be larger when *Setaria surgens* is present in the area (Node 4). Flocks were smallest when the percentage of both grasses was lower (Node 3) (Fig. 3.7). Ground cover variables more strongly split nodes than did woody plant cover variables.

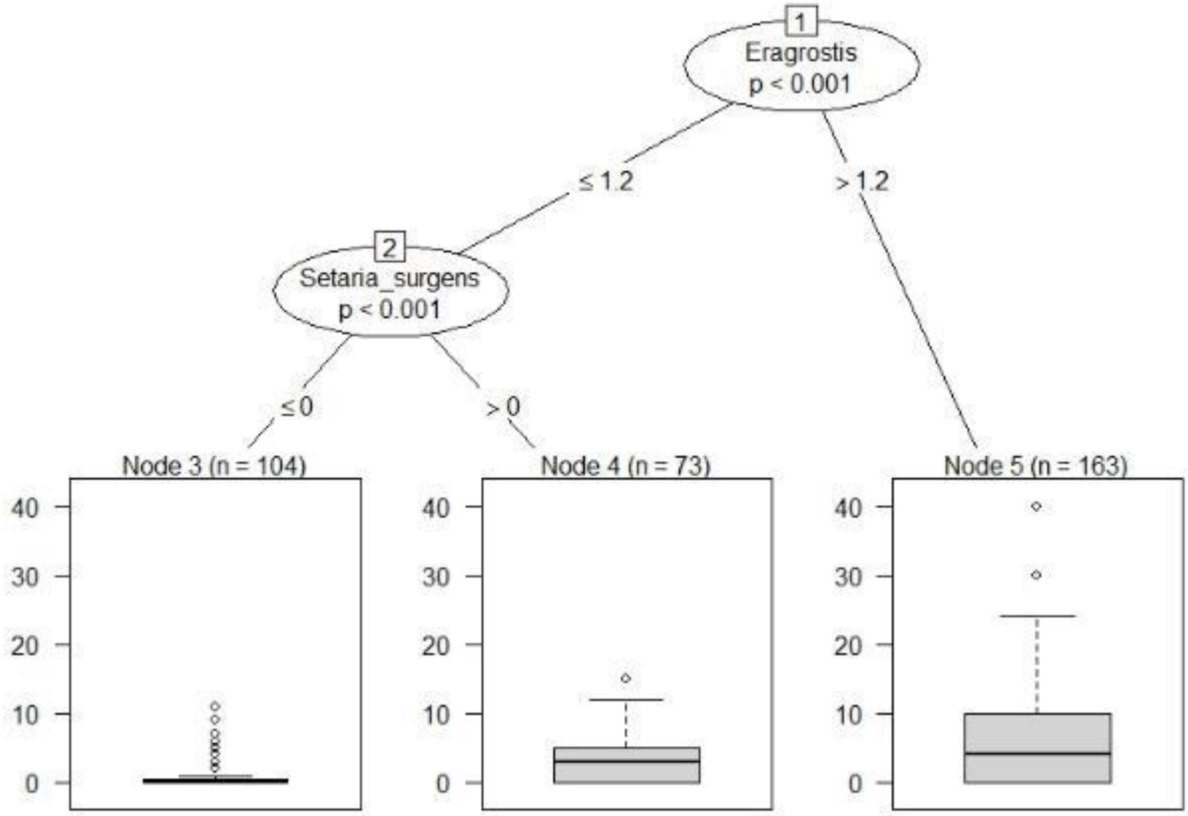

Fig. 3.7. Conditional inference tree examining BTF flock size in different sites using both ground cover and woody plant cover composition as explanatory variables (forbs were excluded; number of variables = 68). Flock size was best classified into three groups (leaves) considering this subset of data. Node 5 is primarily larger flocks, node 4 medium flocks and node 3 small flocks. Node 1: percentage of cover of *Eragrostis* spp. Node 2: percentage of cover of *Setaria surgens*.

All variables together (ground cover and woody plant cover structural and compositional variables; excluding forbs): larger BTF flocks were associated with high percentages of *Eragrostis* spp. cover (Node 5). Medium flocks were associated with high grass diversity (Node 4). Flocks were smallest when the percentage of *Eragrostis* spp. and grass richness were lower (Node 3) (Fig. 3.8).

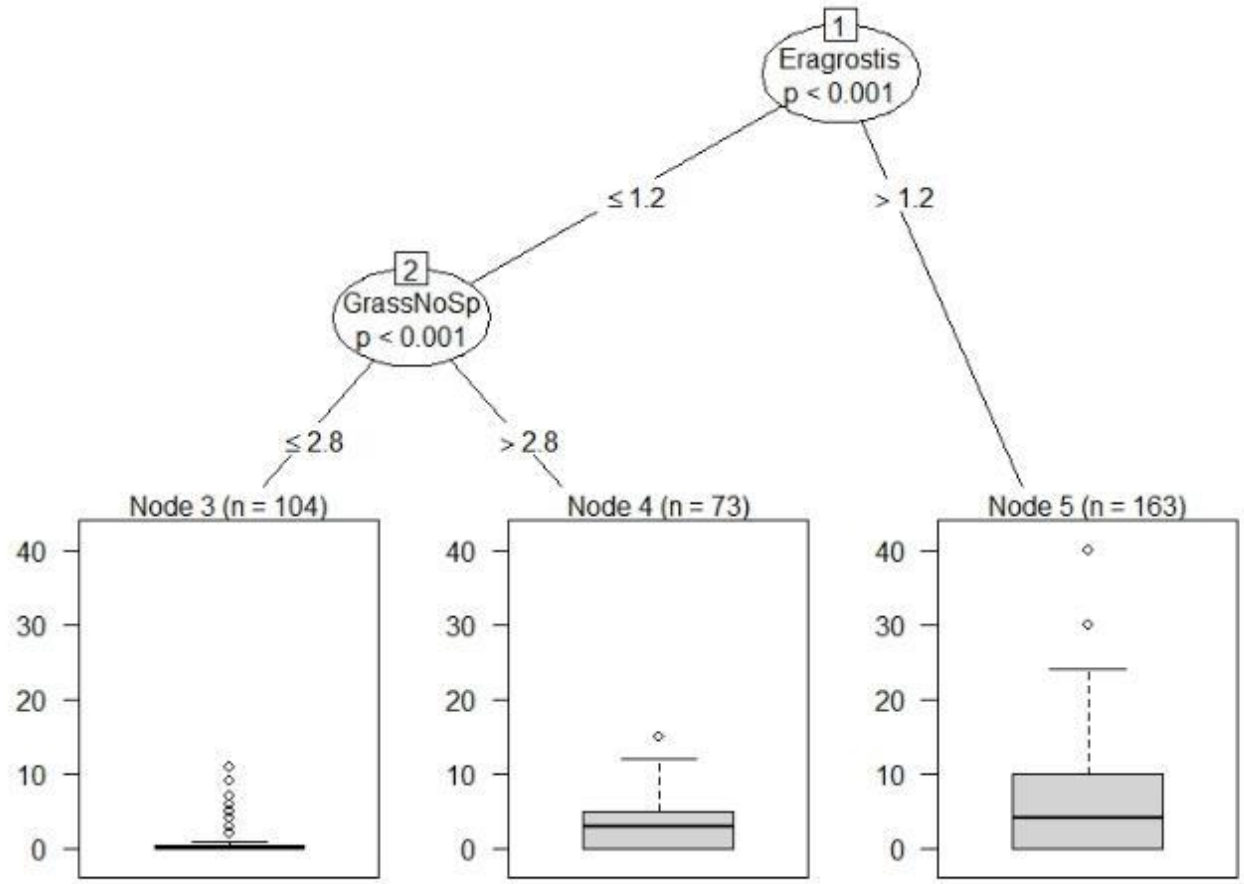

Fig. 3.8. Conditional inference tree examining BTF flock size in different sites using ground cover and woody plant cover composition and structure as explanatory variables (forbs were excluded; number of variables = 110). Flock size was best classified into three groups (leaves) considering all subset of data. Node 5 is primarily larger flocks, node 4 medium flocks and node 3 small flocks. Node 1: percentage of cover of *Eragrostis* spp. Node 2: number of grass species.

Hypothesis testing

Considering the eight hypotheses tested in this study, the ones using variables from grass composition and tree structure explained more of the variation in BTF flock size than models with tree composition variables (Table 3.6). Models 1 to 5 differ little (AIC < 2) in their ability to explain BTF flock size.

The first model (as ranked by ΔAIC; Tables 3.6 and 3.7) refers to the third hypothesis (Table 3.4). This hypothesis is that BTF flocks will be larger when there is high cover of specific grass species (*Chrysopogon fallax, Chloris* spp., *Eragrostis* spp. and *Aristida* *warburgii*). BTF flock size did not change with the percentage of *C. fallax*, *Chloris* spp. or *Eragrostis* spp. but with *A. warburgii*. BTF flock is negatively correlated with coverage of *Chloris* spp. The model partially supports the hypothesis that BTF flocks will be larger with higher abundances of specific grass species in the habitat.

The second model (as ranked by ΔAIC; Tables 3.6 and 3.7) refers to the fourth hypothesis (Table 3.4) that BTF flocks will be larger when the percentages of *C. fallax* and *Eragrostis* spp. are higher, where grass richness is higher (not including forbs) and bare ground is present (quadratic relationship). BTF flock size varied positively with *Eragrostis* spp. (positive relationship) and bare ground, but not with *C. fallax* (negative relationship) or grass richness (negative relationship). The model partially supports the hypothesis that BTF flocks will be larger in areas with high cover of *Eragrostis* spp. and higher grass diversity.

The third model (as ranked by ΔAIC; Tables 3.6 and 3.7) refers to the fifth hypothesis (Table 3.4). The hypothesis is that BTF flocks will be greater when (1) abundances of small, medium and large trees are higher but (2) smaller when shrub abundance is higher. BTF flock size negatively associated with the abundance of shrubs and large trees but positively associated with the number of dead trees standing or the abundance of medium trees. This model partially supports the hypothesis tested because a positive relationship of BTF flocks with abundance of large trees was expected and it didn't happen.

The fourth model (as ranked by ΔAIC; Tables 3.6 and 3.7) refers to the second hypothesis (Table 3.4); that BTF flocks will be larger when vegetation cover and native species cover are higher but visual obstruction and non-native species are smaller. BTF flock sizes varied with all variables but the model does not support the hypothesis in terms of percentage of vegetation cover (showing a negative relationship) or non-native species (showing a positive relationship; Table 3.7). This model also partially supports the hypothesis.

The fifth model (as ranked by ΔAIC; Tables 3.6 and 3.7) refers to the sixth hypothesis (Table 3.4). The hypothesis is that BTF flocks will be larger when abundance of dead trees standing is higher and shrub cover and shrub abundance is lower. BTF flock size varied with dead trees standing (positive relationship) and shrub cover (negative relationship) but not with arboreal abundance.

Table 3.6. Model ranking for habitat requirement preferences of the Black-throated finch southern subspecies (*Poephila cincta cincta*) using negative binomial regression. Models were written a priori and were based on known relevant ecological variables. Hypotheses were first written using variables from different groups (ground cover (1) structure, (2) composition, woody plant cover (3) structure and (4) composition) and then one model considering all variables. Bare ground was considered only with quadratic effect. Also shown are the deviance, number of estimated parameters (K), ΔAIC (difference in AIC value in regard to best model) and AIC weight. Models are ranked according to the ΔAIC.

Model	Group of variables	Model formula	Null Deviance	Residual deviance	AICc	ΔAIC	AIC weight	How much the model explains	K
1	GC	*Chrysopogon fallax* + Chloris + Eragrostis + *Aristida warburgii*	142.482	97.161	585.88	0	0.2381	68%	4
2	GC + GS	*Chrysopogon fallax* + Eragrostis + BG^2 + GrassNºSp	142.363	97.169	585.97	0.09	0.2279	68%	4
3	TS	DTstanding + abundance of i) shrubs + ii) medium trees + iii) large trees	142.293	97.158	586	0.12	0.2237	68%	4
4	GS	VC + VO + %Native + %Exotic	141.219	97.451	587.02	1.14	0.1345	69%	4
5	TS	DTstanding + shrub cover + total arboreal abundance	138.26	97.7	587.3	1.42	0.1169	70%	3
6	GS + TS + TC	BG^2 + %Grass + GrassNºSp + large trees abundance + *Eucalyptus platyphylla* abundance	141.468	97.335	588.74	2.86	0.0570	68%	5
7	TC	*E. platyphylla* + *Melaleuca nervosa* + *Ziziphus mauritiana*	126.544	98.186	596.37	10.49	0.0013	77%	3
8	GS	BG^2 + %Grass + GrassNºSp + %Forb	127.817	98.894	598.1	12.22	0.0005	77%	4
9		Null model	101.76	101.76	615.69	29.81			0

Table 3.7. Models with substantial level of support. Significance: '***' = 0; '**' = 0.001, '*'= 0.01, '.' = 0.05, '' (blank) = 0.1 and '1' if larger than 0.1.

Model	Coefficients	Estimate	Std. Error	z value	P	Significance
GC	Intercept	1.392	0.201	6.921	4.48e-12	***
	Chrysopogon fallax	0.045	0.025	1.790	0.0734	.
	Chloris	-0.037	0.255	-0.145	0.8845	
	Eragrostis	0.180	0.146	1.236	0.2165	
	Aristida warburgii	0.062	0.014	4.511	6.46e-06	***
GC + GS	Intercept	1.768	0.456	3.881	0.0001	***
	Chrysopogon fallax	-0.041	0.038	-1.087	0.277131	
	Eragrostis	0.661	0.180	3.681	0.000232	***
	Bare ground ^{2}	0.000	0.000	-4.473	7.72e-06	***
	GrassNoSp	-0.037	0.084	-0.434	0.6642	
TS	Intercept	2.081	0.227	9.167	<2e-16	***
	DT standing	0.105	0.066	1.601	0.1093	
	SH_N_Ind	-0.010	0.004	-2.365	0.0180	*
	10-40_N_ind	0.019	0.040	0.482	0.6300	
	>40_N_ind	-0.420	0.178	-2.362	0.0182	*
GS	Intercept	-0.315	0.062	-0.509	0.6109	
	LandCover	-0.111	0.031	-3.522	0.0004	***
	VO_mean	0.030	0.005	5.535	3.11e-08	***
	PerNative	0.143	0.032	4.454	8.42e-06	***
	PerExotic	0.122	0.031	3.899	9.67e-05	***
TS	Intercept	1.841	0.168	10.965	<2e-16	***
	Dtstanding	0.146	0.034	4.282	1.85e-05	***
	Line_TOTs_SH	-0.168	0.067	-2.511	0.0121	*
	T_N_Ind	0.002	0.002	0.685	0.4935	

Discussion

BTF favour habitats characterized by the presence of native grass species and standing dead trees. In contrast, habitats with high shrub cover were avoided by BTF. BTF preferred sites with high abundances of *Eragrostis* spp. and high grass diversity. In addition, there is a positive influence of the density of vegetation below 20 cm in height. Compositional and structural features of ground cover influence BTF more than woody plant cover. Although BTF have specific nesting requirements such as tree species (Chapter 4), in this study the results of the compositional features of the landscape did not seem to play an important role. These results represent a major step toward a better understanding of the requirements of BTF, as they expose the main vegetation factors that drive habitat selection, thus providing key information for effective management actions.

Vegetation structural characteristics are a primary determinant for bird communities in Australian tropical savannas (Tassicker et al. 2006) and North America (Rotenberry and Wiens 1980). In this study it was observed that structural features are important specifically for BTF. However, when variables of grass structure and composition were analyzed together (conditional inference trees), the presence of one grass species (*Eragrostis* spp.) was the preeminent variable, followed by *Setaria* spp. Based on the current published BTF management guidelines (BTFRT/NRA 2011), important grasses for the species are pitted bluegrass (*Bothriochloa decipiens*), windmill grass (*Chloris* spp.), love grasses (*Eragrostis* spp.), panic grasses (*Panicum* spp.), fairy grass (*Sporobolus caroli*), annual wanderrie grass (*Eriachne armittii*) and pigeon grasses (*Setaria apiculata* and *Setaria surgens*). Therefore the findings of this study on Townsville Coastal Plain corroborate the indications of the management guidelines for BTF habitat in the Northern Brigalow Belt. While *Aristida warburgii* has been considered a species that BTF avoid (BTFRT/NRA 2011, Buosi 2011), the opposite was observed in this study. This might be a matter of *A. warburgii* being the most abundant grass species at the site with the largest flock of BTF recorded during this study and the low number of sites surveyed; or the birds might have actually been using this resource. Accordingly to the management guidelines, BTF tend to avoid black speargrass (*Heteropogon contortus*) and kangaroo grass (*Themeda triandra*); also *Heteropogon contortus*, although not included in our models, was the only grass species present at all the study sites; while *A. warburgii* and *T. triandra* were the species most abundant on those two sites where BTF flocks were largest. Birds might be using those resources or simply using the sites in spite of the high abundance of these grasses. Further studies on diet should be undertaken to elucidate the roles of specific grass species in the ecology of BTF. The monitoring of areas where BTF flocks were and were not commonly seen is also important to verify if those grasses are being used as resources or if they are negatively affecting the birds.

Non-native and invasive species negatively impact many native flora and fauna, affecting ecosystem functions, species distribution and population dynamics of native species (Fleishman et al. 2003). However, the opposite was observed in this study, with BTF flock size positively related to specific non-native grasses. This could be related to the fact that some of those non-native grasses are a useful food source to BTF on the Townsville Coastal Plain. For example, *Urochloa mosambicensis* dominated BTF diet during the non-breeding period, and *Digitaria ciliaris* dominated diet during breeding period (Mitchell 1996). The management guidelines for BTF do emphasize the control of weeds as an important action to restore its habitat. Although not shown in the models or conditional inference trees, some weeds, for instance *Themeda quadrivalvis* and *Stylolanthes scabra*, act negatively upon foraging strategies of the birds (pers. observations). During this study, sites where *T. quadrivalvis* increased in abundance coincided with a decline in BTF flock size; and sites where small patches of *S. scabra* were burnt were used by the birds for foraging. While some non-native species are used by BTF, others have a negative effect on the local population. *T.*

quadrivalvis is a major weed in Queensland (Keir and Vogler 2006) and invasive grasses are known to impact wildlife, particularly granivorous birds as the seeds of some species can be less acceptable to the birds (Grice et al. 2013), can have high biomass and, in case of BTF, affect the accessibility of seed to ground-foraging birds. The threat to wildlife from non-native plants is known worldwide (Brooks et al. 2004). Eradication of particular plant species and restoration of habitat is the preferred action, although it is ecologically and economically challenging (Fleishman et al. 2003). Efforts to control weeds in the vicinity of Lake Ross have had little impact; *T. quadrivalvis* infestations increase with fire and overstocking and is considered one of the most prevalent weed in rehabilitation projects done by local organizations (TCC 2000). To maintain a sustainable population of BTF it is imperative to understand how particular non-native plants are affecting them and their habitats.

While the species may benefit from some exotic grasses, BTF avoided sites with high shrub coverage and abundance, particularly of Indian jujube (*Ziziphus mauritiana*), lantana (*Lantana camara*) and Townsville wattle (*Acacia leptostachya*). Additionally, the total number of trees with DBH greater than 40 cm had a negative relationship with BTF flock size. Shrub proliferation – the increase in woody plant biomass in savanna landscapes leading to a transition from grassy to shrubby ecosystems – is considered one factor contributing to changes in fauna guilds in this ecosystem (Roques et al. 2001, Tassicker et al. 2006). Shrub proliferation in the world's savannas and grasslands benefits some species but disadvantages others (Tassicker et al. 2006, Sirami et al. 2009, Kutt and Martin 2010, Sirami and Monadjem 2012). As the habitat become more shrubby, grass production typically declines dramatically (Scholes and Archer 1997), leading to a decline in seed availability with consequences for the granivore assemblage. In Australia, shrub proliferation is associated with the declines of the Golden-shouldered parrot *Psephotus chrysopterygius* on Cape York Peninsula (Crowley and Garnett 1999). Woinarski and Ash (2002) related the low abundance of granivorous species observed in grazed sites in the Townsville region to the high density of woody weeds (*Lantana camara*) (Woinarski and Ash 2002). Shrub cover in our study involves mainly *Acacia leptostachya*, *Lantana camara* and *Ziziphus mauritiana,* as those species together represent 85% of total abundance of all species classified within this lifeform.

Dead standing trees were found to be an important structural feature in BTF habitat. Dead standing trees were observed being used by BTF in two situations: to rest during a long flight (if the flock was flying a distance greater than 200 m the birds would stop on higher branches, usually greater than 4 m height, before continuing the flight); and standing dead trees – about 2 m height – were frequently used by BTF during foraging bouts as an intermediate stratum to access the ground. Sites with a higher abundance of dead trees have been shown to be important for bird populations across the world, supporting more diverse bird assemblages and providing important habitat for declining bird species (Gibbs et al. 1993, Styring and Ickes 2001, Moreira et al. 2003). Despite the known importance of dead trees as a structural feature for birds, our study was the first to document this for BTF. Although the shorter dead trees appeared to be important for BTF flocks it is important not to confound this with a shrubby environment, which had a negative effect on flock size in our study. The potential for BTF to be detected more readily while using dead trees should be acknowledged, though the association between BTF and dead trees was noted throughout this study (Chapters 2, 3 and 4), and while conducting the radio-tracking surveys I was actively following BTF and observing their foraging behavior, therefore diminishing any biased in the sampling. Another important tree structural characteristic was the presence of *Acacia holosericea*; this species possibly has the same structural importance in BTF habitat as shorter dead standing trees. BTF were observed using these as a medium stratum to easily access the ground. It was also observed that some individual BTF perched on those structures while the rest of the flock was foraging, so perhaps those structures are used to increase vigilance during foraging bouts, as is common in several flocking species (Elgar 1989, Bekoff 1995, Whittingham et al. 2004, Butler et al. 2005a). (Elgar 1989, Bekoff 1995, Whittingham et al. 2004, Butler et al. 2005a). In this study, dead trees were apparently killed by fire (prescribed or non-prescribed as lightings) or storms, so these processes will also affect the suitability of BTF habitat.

BTF appeared to prefer sites with higher densities of grass up to 20 cm high. This might indicate that BTF flocks are also selecting habitat based on vegetation height and visual obstruction. Vegetation height will affect how granivorous birds are using the landscape (Butler et al. 2005b) as this assemblage might benefit in habitats with less visual obstruction. Vegetation height has been shown to affect some granivorous birds (e.g. Yellowhammer *Emberiza citrinella*, Greenfinch *Carduelis chloris*) (Whittingham et al. 2009) because they benefit from a lower perceived predation risk in a shorter vegetation as they can detect predators more efficiently. Additionally, the fact that BTF forage on the ground rather than on grass stems, small patchy areas with bare ground and less dense vegetation will benefit birds by allowing access to the seed bank (Chapter 5). In this study, BTF flocks had a positive relationship with bare ground. However, a combination of different factors such as the percentage of native species, total number of grass species and bare ground are more important than bare ground alone. Annual grasses are also believed to play an important role in BTF diet in the dry season (Garnett et al. 2011a).

Although most of the known reliable sites for BTF near Townsville were surveyed in this study, the relatively low total number of sites (10) may mean that one or two specific sites could be influencing the overall results. Additionally, bird density can be misleading when assessing habitat preferences as birds may occupy low-quality habitats, leading to maladaptive habitat selection (Fuller 2012b). Nonetheless, it is still a reasonable indicator for habitat quality (Bock and Jones 2004, Pérot and Villard 2009), especially considering the other ways of measuring habitat quality (clutch size, population age structure, survival rate, and reproductive success among others). The radio-tracking data (Chapter 2) indicate that larger BTF flocks were sighted near foraging areas and water sources; the nesting areas were formed only with single nests or loose colonies and all sites were monitored all year round. Thus, all variation in flock size during the year were covered by the bird surveys and flock size in this study is a reasonable indicator of basic ecological requirements of BTF. Therefore, the results of this study represent an appropriate first-step approach to understanding the requirements of an endangered, under-studied species. Further studies could target a larger number of sites, but focusing on specific variables that this study has identified. Other factors that might be considered include the seasonality of savanna landscapes. Despite the logistical difficulties of gathering robust field data, particularly in the wet season, information from all seasons may provide a more complete picture of BTF habitat requirements.

All sites in this study were surveyed in the dry season and this might have had an effect on our results. Savannas landscapes, particularly ground cover parameters, can be very different between wet and dry seasons. Seed abundance and vegetation density, for instance, will decrease over the course of the dry season. Changes in landscape structure will affect granivorous birds as it will affect not only availability of seed but also accessibility of seeds. This study points to relevant features of the habitat in the dry season and future studies should investigate seasonal nuances in habitat preferences. Bird sampling, although carried out throughout the year, was also more effective in the dry season. BTF have very cryptic behaviour and sightings during the wet season were scarce.

Some areas where BTF are persisting are highly modified (grazed, overgrazed, weed invaded, eroded or used intensively for recreation purposes) whereas other places, that appear reasonably intact, are apparently not used by the birds. Based on the data presented here and on field observations, BTF requirements involve different aspects of ground cover and woody plant cover. Overall they feed on a variety of grasses (native or non-native) and structurally they seem to prefer areas with a mosaic of vegetation features. Within the areas they occupy, features such as low vegetation density are relevant. The presence of grasses is vital to provide food, but low vegetation density will allow them to access those seeds. The mosaic area (hundreds of metres) should include the presence of specific grasses (abundance of good grasses rather than only high richness) in proximity to patches of bare ground and low vegetation density. Field observations indicated that sites that were not managed (no patchy fires or controlled grazing) usually have a high biomass and ground cover density, and birds were never seen foraging on those areas. Extensive overgrazed areas or eroded areas were either not used by BTF or only visited to obtain water. It appears that extensive homogeneous areas (overgrazed with no ground vegetation at all or unmanaged and with dense ground vegetation) does not meet the BTF requirements. It was not within the scope of this study to measure the level of grazing or burning intensity but it was apparent that the birds need a heterogeneous habitat, where food availability and accessibility are key factors. Other important features that need to be considered in the mosaic are the woody plant cover. Shrubby environments seem to negatively affect BTF; control of *Z. mauritiana*, *L. camara* and *S. scabra* will improve the quality of the habitat. In this chapter no arboreal species (woody plant cover composition) showed particular relevance to BTF habitat. Nonetheless a structural feature that seems to improve the habitat for the birds is the presence of a medium strata feature, either a dead tree or a fine shrub such as *A. holocericea*, which will be used by BTF to access the ground. This event was observed many times in the field: a tiny vertical stick near a patch of bare ground close to grasses was heavily used by BTF. The positive relationship with the presence of dead trees should not be confounded with the density. Shrubby areas are usually extensive areas and reduce the birds' access to food, while the medium strata structure tends to have low density, not dominate the landscape and is used by the birds to access foraging patches. The mosaic should also include a water source and specific trees for nesting sites (Chapter 4).

Chapter 4

Nesting tree characteristics and nest site selection of the Black-throated finch southern subspecies, *Poephila cincta cincta*, in north-eastern Australia [3]

Abstract

Declines in granivorous bird assemblage have been reported throughout Australia. Understanding how this bird assemblage uses the landscape and identifying their ecological requirements is a prerequisite for successful conservation outcomes. Nest site selection studies provide relevant ecological information which can support conservation and management actions. The Black-throated finch southern subspecies (*Poephila cincta cincta*) is a granivore endemic to eastern Australia. It is listed as endangered under State and Commonwealth legislation and, despite the rapid habitat loss and range contraction, little is known about its ecological requirements. This study aimed to describe the most relevant characteristics of nesting trees and investigate if the birds are selecting particular features for nesting habitat compared with the available habitat (breeding and roosting nests were used in this study). Surveys were conducted in north-eastern Australia, between 2011 and 2014. Common woodland species *Eucalyptus platyphylla* and *Melaleuca viridiflora* were the preferred nest trees. Nesting locations had lower diversity of ground and woody plant cover compared with available habitat. There was no selection by BTF of nesting locations with particular ground cover parameters but they showed a preference for locations with low tree abundance ($\bar{x}$ = 641.9 ± 180.8). Abundance of the trees *Corymbia clarksoniana*, *E. drepanophylla*, *E. platyphylla* and *Planchonia careya* was considerably lower in used habitat. In the predictive models, nest site selection by BTF was best explained by density of *E. drepanophylla* and the Shannon Diversity Index for grasses. The best predictive model of BTF nesting habitat showed that BTF nesting areas are positively correlated with percentage of the grasses *Heteropogon contortus* and *Themeda triandra,* the shrub *S. scabra* and negatively correlated with densities of *E. drepanophylla*. BTF are particularly vulnerable to increases in tree density, shrub proliferation and a reduction in grass diversity.

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[3] This chapter was submitted as a paper:
Rechetelo, J., Jones, R., Reside, A.E., Hardesty, B.D., Moloney, J. and Grice, A.C. 2015. Nesting tree characteristics and nest site selection of the Black-throated finch southern subspecies, *Poephila cincta cincta*, in north-eastern Australia.

Introduction

Successful conservation outcomes rely upon our understanding of how birds are using the habitat and their specific requirements (Fuller 2012a). Nesting, foraging and roosting activities require environmental resources the availability of which can vary between seasons and locations; e.g. general characteristics of areas used for nesting might differ substantially from those of areas used for foraging (Block and Brennan 1993). Birds distinguish between various environmental components and disproportionally use a particular habitat, for instance the nest sites, with consequences for fitness and survival (Block and Brennan 1993, Jones 2001, Goodnow and Reitsma 2011). Selection of a nest site may be affected by factors such as vegetation type or predator distribution at different scales (Martin 1993b, a, Citta and Lindberg 2007, Barrientos et al. 2009). Habitat structure is often used as an indicator of habitat quality (Martin 1998, Thogmartin 1999, Warren and Anderson 2005, Murray and Best 2014). However, nest site selection for colonial or social species may depend upon other factors; for example, individuals might use the presence of a conspecific – or conspecific nesting density – as a cue for habitat suitability (Mariette and Griffith 2012).

Nest site selection studies are required to provide ecological information that can direct conservation actions (Morán-López et al. 2006, Pasinelli 2007, Rocha et al. 2013). Conservation actions targeting granivorous birds are paramount in Australia as broad-scale declines in granivorous bird assemblages are occurring in many regions. Although these declines are well documented (Franklin 1999, Franklin et al. 2005), for some declining species there is a lack of knowledge of specific ecological requirements (Garnett and Crowley 2000c, Higgins et al. 2006a).

The Black-throated finch (*Poephila cincta*) is a small granivore endemic to north-eastern Australia. It inhabits open, dry, grassy woodlands dominated by eucalypts, *Acacia* spp. or *Melaleuca* spp. (Zann 1976a). Historically its range stretched from north-east New

South Wales to northern Queensland (Keast 1958, Higgins et al. 2006a, BirdLife International 2015a). Although changes in land use have led to a severe range reduction (Garnett and Crowley 2000c, Higgins et al. 2006a), little is known about its ecological requirements (Buosi 2011).

The range of the Black-throated finch southern subspecies (*Poephila cincta cincta;* herein referred as BTF) has contracted by around 80% since the 1970s (BTFRT 2007a, Department of the Environment 2015a). The decline in the southern parts of the species' range started in the early 20th century and by the 1940s they had mostly disappeared from the south-east of their former range (Buosi 2011). BTF is now patchily distributed and it is listed as endangered under the Commonwealth Environment Protection and Biodiversity Conservation Act (EPBC 1999), the Queensland Nature Conservation Act (Queensland Government 2006a) and the New South Wales Threatened Species Conservation Act (New South Wales Government 1995b). Clearing, fragmentation, habitat degradation (including changes in vegetation structure and composition), alteration in fire regimes, and introduced plants and animals are among the perceived threats to the species (Garnett et al. 2011a).

Although there have been ecological studies focusing on BTF (Zann 1976a, Zann 1977, Mitchell 1996, Isles 2007b, Whatmough 2010, Maute 2011), little information exists on nesting ecology and nesting site selection. Information on nesting ecology is an important first step in identifying suitable habitat and establishing effective recovery programs (Fessl et al. 2011), and has been highlighted as a priority action for BTF (Garnett et al. 2011a). The earliest published observations of BTF nesting areas date from 1976-1977 (Zann 1976a, Zann 1977). They form loose colonies, and use nests for breeding and roosting (Zann 1976a, Isles 2007b, Buosi 2011). The nest is domed and made of grass in the foliage of trees or in tree hollows, laying 5-9 eggs (Immelmann and Cayley 1982). Clutch size is 5.1 ± 0.9 clutches per nest and incubation time was 14.6 ± 1.7 days (aviary birds(Zann 1976a)). The breeding period is from October to April, but depends on the season, location and resource availability (BTFRT 2007a, BTFRT/NRA 2011, Buosi 2011). Observations suggest that this species has breeding site fidelity (returning to the same area every breeding season) and selecting nesting areas based on resource availability (Isles 2007b). However, no strong, evidence-based information about nesting sites is yet available.

Resource availability is likely to be a key driver of choice of nesting areas; a combination of water source proximity and the presence of suitable foraging areas near the nesting colony is likely to influence their choices of nesting habitat (Isles 2007b). Therefore, in this chapter, I investigate the microhabitat parameters associated with nesting sites of Black-throated finch southern subspecies on the Townsville Coastal Plain, Queensland, Australia, aiming to identify and describe the important characteristics of BTF nesting site.

Methods

Study area

Surveys were conducted in the vicinity of Lake Ross, south of Townsville, Queensland, in north eastern Australia during 2011-2014. This area supports a substantial and readily accessible BTF population (BTFRT/NRA 2011). The vegetation consists predominantly of eucalypt open woodland and *Melaleuca* low open woodland (Murtha 1975). The vegetation is significantly modified by human activity throughout the study area, much being cleared or partially cleared. Virtually all areas have invasive grasses and shrubs occurring in scattered clumps or in extensive infestations (detailed description in Chapter 1).

Data collection

BTF build domed nests of grass in the foliage of trees or in tree hollows (Immelmann and Cayley 1982). Nests can be identified by careful visual inspection of trees and bird behaviour. Nests of BTF were located by random searches of the study area, information from a public network (BirdLife Townsville members, personal communication) and by following radio-tracked individuals (Chapter 2). Only active nests were recorded (birds sighted entering the nest, carrying nesting material or roosting). There is no evidence that BTF use different nests to breed, rear juveniles or roost and field observations suggest that they may breed and roost in the same nest, so all nests will be used in the analysis. The position of each BTF nest was marked using a global positioning system (Model GPSmap62s, Garmin).

For 50 nests I recorded 10 variables potentially important in characterising the habitat (Table 4.1) and for a random subset of 20 nests also recorded information about ground cover and woody plant cover around the nest tree (Table 4.2 and Table 4.3). Ground cover parameters, including vegetation structure and composition of grasses, forbs, sedges and herbs, were assessed by randomly placing a 0.25m^2 quadrat anywhere within 5m radius of the nesting tree, having the nesting tree in the centre (Table 4.2; Fig. 4.1). To examine the woody plant cover (structural and compositional measures of trees, shrubs and vines; Table 4.3; Fig. 4.1) arboreal abundance (trees and shrubs) individuals were included (Martin 1998, Oppel et al. 2004), and a larger (11m radius) was used. The larger area was to better represent the presence of arboreal species around nesting trees, while the 5m radius circle restricted the sampling of the ground cover variables to close to the nesting tree. Areas around nesting trees with a circle of 5m radius nested within a circle of 11m radius (Martin 1998, Oppel et al. 2004) will be referred to as ‘nesting locations’ and will be treated as ‘used habitat’ (Fig. 4.1). A circle of larger radius was used to sample woody cover parameters to better represent the nest surroundings in savannas landscape where the trees are spread out. The species *Stylosanthes scabra*, although a shrub, was included in the ground cover survey along with grasses and forbs due to 1) its abundance in the surveyed areas, 2) because it is an introduced pasture plant that can replace native grasses (and reduce foraging efficiency of BTF) 3) and to better represent the species in the surveyed areas.

Table 4.1. Variables recorded from individual nesting trees within the study area (N = 50).

Variable	Description
Nest location	Nest location: (a) in tree hollow (b) in tree foliage
DBH	Diameter breast height. Group in 3 classes: small trees (DBH less than 10cm), medium trees (DBH between 10 and 40 cm) and large trees (DBH larger than 40cm)
Nest height	Height of the nest (above ground). Visual estimation in meters. Classes: Low (nest less than 5m high), Medium (6-10m) and High (≥11m)
Tree species	Species of nesting tree
Tree height	Height of nesting tree. Visual estimation in meters. Classes: Short trees (smaller than 5m height), medium trees (6-15m) and tall trees (≥16m)
Mistletoe	If the nest was built within mistletoe plant
Distance to Water	Distance to the closest water (creek or dam); measured in GIS using the coordinate from the nest and the position of the watering point (m). Classes: (1) <400m, (2) 400-1000m and (3) >1000m
Distance to the Nearest Nest	Distance to the closest nest; measured in GIS using the coordinate from the nest and the position of the watering point (m). Classes: (1) <100m, (2) 100-400 and (3) 400->1000
Flock size	Number of BTF using the nest (e.g. carrying nest material, roosting, etc (when possible)). Classes: pairs (only 2 birds), trios (3 birds), small group (4 birds) and large groups (5 birds or more)
Season	Season (wet or dry) when the nest was found

To check if BTF were selecting particular features in the habitat I measured the same variables at nesting locations (used habitat) and in transects randomly placed across the study area (available habitat). Ten areas were surveyed using the methodology BioCondition (Eyre et al. 2011), whereby five transects of 100m each were placed at each area, assessing the same variables for ground and woody plant cover (Table 4.2 and Table 4.3). Five 1m^2 quadrats were placed along transects to describe ground cover; while woody plant cover information was collected along each transect in a 10m wide area using a tape measure (Fig. 4.1; for more details see Chapter 2). Other variables including vegetation density and visual obstruction were also assessed based on their relevance in grassy habitats (Wiens 1969, Robel et al. 1970, Fisher and Davis 2010). Vegetation density was assessed by counting the number of hits of leaves/stems using a 1m pole (Table 4.2). Vegetation density was estimated using three quadrats in used habitats and five quadrats in available habitat and means calculated for each. The variables were grouped as: ground cover structure, ground cover composition, woody plant cover structure and woody plant cover composition (Table 4.2 and Table 4.3).

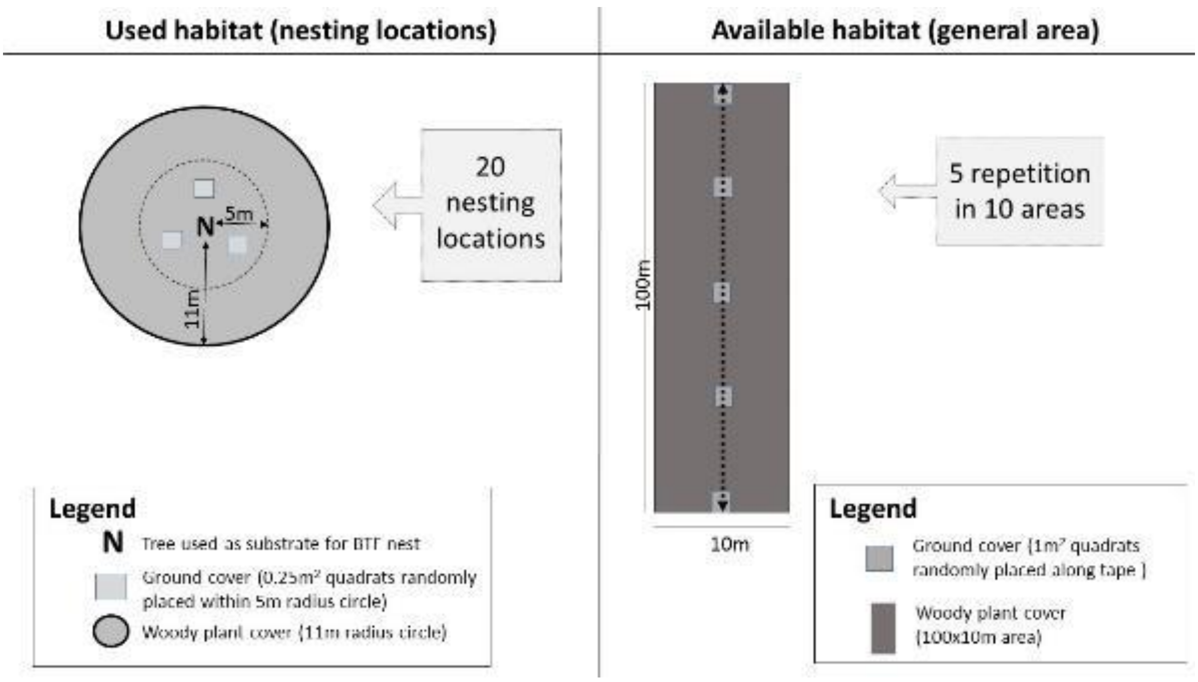

Fig. 4.1. Sampling design for vegetation surveys in used and available habitat.

Table 4.2. Ground cover environmental variables measured for BTF nesting locations (used habitat: variables recorded in each quadrat of a subset of 20 nests) and general areas (available habitat: variables recorded in 5 transects x10 areas around of Ross river dam).

Variables	**Definition**		
Ground Cover Structure	GCD		Ground cover (vegetation) density: n leaves/stems intercepted at a vertical point recorded in 20 cm height increments from ground level. Six classes were analysed based on the stratum of the vegetation: GCD 0-20cm, GCD 20-40, GCD 40-60, GCD 60-80, GCD 80-100 and GCD >100cm. Unit: counts of ‘hits’. Mean was calculated for each class.
	VO		Visual obstruction – a 1m pole was placed in the centre of the quadrat and an observer estimated the visual obstruction from an eye height of 1.0m and 4m from the pole (e.g. in an area with no ground cover, 100% of the pole was observed and this was the visibility recorded). VO was estimated 3 times and the mean was calculated.
	FWM		Fallen woody material – Count of logs > 10 cm diameter and 0.5 m long in an area 50 x 10 m along the tape measure
	Total 100%	BG	% of bare ground in a 1m^2 quadrat (BG, rock, litter and ground cover added 100%)
		Rock	% of rock (as above)
		Litter	% of litter (as above)
		VC	% of vegetation – grasses, forbs, seedlings (as above)
Woody Plant Cover Composition			Grasses, forbs and sedges identified to species where possible, otherwise genus, or family. Life form and status (native, introduced) recorded

Table 4.3. Woody plant cover (tree/shrubs/vines) environmental variables measured for BTF nesting locations (used habitat: variables recorded in a subset of 20 nests) and general areas (available habitat: variables recorded in 5 transects x10 areas around of Ross river dam).

Variables		**Definition**
Woody Plant Cover Structure	Shrub abundance	*n* individual shrubs per hectare (calculated for the11m radius around nesting trees and in a $100x10m^2$ area along transects). Counts converted in hectare.
	Shrub richness	*n* shrubs species (as above)
	Small trees abundance	*n* individual trees DBH < 10 cm / ha
	Small trees richness	*n* tree species DBH <10 cm / ha
	Medium trees abundance	*n* individual trees DBH 10-40 cm /ha
	Medium trees richness	*n* tree species DBH 10-40 cm /ha
	Large trees abundance	*n* individual trees DBH > 40 cm /ha
	Large trees richness	*n* tree species DBH > 40 cm /ha
	Total arboreal abundance	Total *n* trees and shrubs individuals /ha
	Total arboreal richness	Total *n* tree and shrub species/ ha
	Basal area	Basal area measured in the centre of each transect (50m mark) by using cruise master prism
	DTS	*n* dead trees that were still standing and used as perch/ hectare
	DTG	*n* dead trees on the ground
Woody Plant Cover Composition		Trees, shrubs and vines were identified to species where possible, though some could only be assigned to genera

Data analysis

I calculated descriptive statistics (mean and median) for the variables collected at 50 BTF nest locations. I used linear regression to test the relationship between nest height and tree height. Logistic regression and analysis of deviance was used to identify parameters related to nest tree preference. Only one species of shrub, the introduced Indian jujube (*Ziziphus mauritiana*), was used in the analysis of nest tree selection as this shrub is described in the literature as a potential nest site (Higgins et al. 2006a). Therefore, of the 31 tree species and 11 shrubs in the available habitat, 32 species were used for the nest tree selection analysis. The nest tree preference was measured as the proportion of trees with a nest for each species which had any nests.

To evaluate nesting site selection, I calculated mean and median for the different variables (ground cover structure and woody plant cover structure). I used Welch's two sample t-test to compare variables (ground cover and woody plant cover) in used and available habitats to identify any differences in nesting locations. All structural variables from ground cover and woody plant cover groups were compared, but grass/forb/sedges and tree/shrub composition variables were not included as some of them were found in low numbers during the surveys. Density (plants per ha) was used for each species in order to compare trees in used and available habitats. Tree density was calculated for species in which there were 15 or more individuals in the total counts (used locations plus general areas) plus *Ziziphus mauritiana*. I compared the density of those selected species of trees and *Z. mauritiana* per ha at used and available sites using Welch's t-test. A descriptive assessment of ground cover and woody plant cover composition was made. I calculated the Shannon diversity index for ground cover composition and woody plant cover composition in used and available habitats and compared percentage of grasses in used and available habitat only for the species that were present in nesting locations.

Variables that came out as significantly different between used versus available habitat were used in logistic regression. For the logistic regression (fitted with the R function gIm; family binomial; binary response) to distinguish used from available habitats; the relative importance of the variables was assessed by fitting all possible combinations of the selected variables using the function dredge in the MuMIn package (Bartoń 2015) to find the best predictive model for nesting sites. Akaike Information Criterion (AIC) is a method for selecting a model from a set of models (Burnham and Anderson 2002). In this study, AICc was used instead of AIC, as the number of included parameters was high in relation to sample size as this method corrects for it (Burnham and Anderson 2002, Barton and Barton 2014, Bartoń 2015). Model selection was performed by using the model with the highest weighted AICc value. Akaike weights give the probability that a model is the best model, given the data and the set of candidate models (Burnham and Anderson 2002). The selection for grass species to include in the model were based only on species positively or negatively associated with BTF (Mitchell 1996, BTFRT/NRA 2011) and field observations (Rechetelo et al.; Chapter 2), being those species that were reported as either important habitat or species that BTF specifically avoid. All analysis were performed with the statistical package R version 3.1.1 (R Core Team 2014) and packages 'lattice' (Sarkar 2015) and 'car' (Fox and Weisberg 2011).

Results

Nesting tree parameters

Fifty currently occupied BTF nests were found at seven locations out of 10 monitored during this study. A total of 26 nests were in *Eucalyptus platyphylla*, 14 in *Melaleuca viridiflora*, seven in *Corymbia tessellaris* and three in *Corymbia dallachiana*. Considering all 32 species recorded in the available habitat, BTF prefer to nest on *Eucalyptus platyphylla* (z =

-16.916; df = 31, $p < 0.0001$) and *Melaleuca viridiflora* ($z = -4.233$; df = 31; $p < 0.0001$). Nest height positively correlated with tree height, with nests being placed in the top quarter of the tree ($F_{1,49} = 602.4$, $P < 0.0001$; $y = 0.7x$; explaining 92% of the total variation in nest height). Forty-nine nests were dome-shaped and placed in the foliage of trees and one was inside a hollow branch. Considering DBH, 50% of the nests were in medium trees (Table 4.4). Mean height of nests was 6.5 m while mean tree height was 9.0 m (Table 4.4).

Table 4.4. Height (m) of BTF nests, nesting trees and DBH of nesting trees (classes low, medium and high). Distance to the water and to the nearest nest (classes 1, 2 and 3) (N=50). Classes: nest height low (nest < 5 m high), medium (6-10 m) and high (≥ 11 m); tree height: low - short trees (< 5 m height), medium trees (6 - 15 m) and height - tall trees (≥ 16 m); DBH: low - small trees (DBH < 10 cm), medium trees (DBH 10 - 40 cm) and high - large trees (DBH > 40 cm); distance to the water: (1) < 400 m, (2) 400 - 1000 m and (3) >1000 m; distance to nearest nest: (1) < 100 m, (2) 100-400 m and (3) 400 > 1000 m.

	Mean	Min	Max	*n* low (%) / 1	*n* med (%) / 2	*n* high (%) / 3
Nest height	6.5	1.9	15.0	20 (40)	26 (52)	4 (8)
Tree height	9.0	2.0	17.0	15 (30)	32 (64)	3 (6)
Three DBH	-	-	-	17 (34)	25 (50)	8 (16)
Distance to the water	324.9	5.0	1420	32 (64)	17 (34)	1 (2)
Distance to nearest nest	71.2	2.0	2920	35 (70)	11 (22)	4 (8)

Nests in mistletoe made up 16.3% of all nests, all located in three of the 10 sites. Just under half (n = 21) of the nests were found in the dry season, and 29 in the wet season. Sixty-four percent of all nests were within 400 m of a water source and 70% of all nests were less than 100 m from the nearest nest (Table 4.4).

Nesting site selection

Of the ground cover structural characteristics, only fallen woody material differed significantly (lower in used areas) between used and available areas (Table 4.5). A total of 58 grasses or forbs were recorded in the available habitat: 26 species of grasses (five annuals, 15 perennials and six facultative perennials; 13 Australian natives, nine exotic and four unknown) and 32 forb species (nine annuals, 12 perennial and 10 annual-perennial; 16 Australian natives, 11 exotic and four unknown). From the 58 species of grass, forb and sedge recorded, 16 were present in used areas (Appendix C, Table C.1; different areas search could account for the difference). The Shannon Diversity Index for ground cover composition (grasses, forbs and sedges) was significantly lower ($t = 4.319$, $df = 33.871$, $p < 0.001$) in used sites than in the overall available habitat.

Among woody plant cover characteristics, nesting areas contained lower numbers of small, medium and large trees (Figure 4.2; Table 4.6). The other five parameters did not differ (Table 4.6). Although not significantly different, abundance of shrubs, arboreal abundance and basal area (Figure 4.2) were smaller in used habitat. Shannon Diversity Index for trees (trees, shrubs and vines) was significantly different ($t = 2.836$, $df = 34.626$, $p = 0.008$) between used and available habitat, being lower in used habitat.

Table 4.5. Parameters of ground cover structure (grasses, forbs and sedges structural information) compared between nesting (used) and general (available) habitats of Black-throated finch (N=20 random nests and 5 transects in 10 areas for available habitat). Codes: GCD – ground cover density (six classes); VO – visual obstruction; BG – bare ground, VC – vegetation cover and FWM – fallen wood material.

Parameters		**Used**					**Available**					**t value**	**df**	**p-value**
		$\bar{x}$	SD	SE	Min	Max	$\bar{x}$	SD	SE	Min	Max			
GCD (counts – nº of hits)	GCD 0-20	9.6	7.6	1.7	0.3	33.3	6.9	5.4	0.8	0.2	25.8	-1.430	27.01	0.164
	GCD 20-40	4.5	4.8	1.1	0	16.7	2.8	2.7	0.4	0	9.8	-1.514	23.837	0.143
	GCD 40-60	2.3	2.5	0.5	0	8.3	1.8	2.1	0.3	0	9.2	-0.773	29.857	0.445
	GCD 60-80	1.3	1.5	0.3	0	4	0.9	1.3	0.2	0	5.8	-0.885	31.48	0.383
	GCD 80-100	0.6	1.0	0.2	0	3.3	0.4	0.6	0.1	0	2.4	-0.711	24.153	0.484
	GCD >100	0.1	0.2	0.05	0	1.0	1.7	7.5	1.1	0	44.6	1.517	49.258	0.136
VO (%)		70.1	26.4	5.9	19.6	100	70.0	22.2	3.1	19.8	100	0.009	30.283	0.993
Total 100%	BG (%)	20.65	15.28	3.4	0	51.7	24.4	17.6	2.5	0.4	78	0.887	40.132	0.381
	Litter (%)	51.1	17.6	3.9	17	83.3	45.8	16.4	2.3	16	85.4	-1.145	32.867	0.260
	Rock (%)	0.3	0.8	0.2	0	3	0.9	2.1	0.3	0	13	1.716	67.643	0.091
	VC (%)	28.1	15.1	3.4	3.3	61	28.9	16.5	2.3	1.8	65	0.219	38.055	0.828
FWM ha^{-1}		40.7	41.3	9.2	0	131.5	80.4	114.3	16.2	0	540.0	2.128	67.636	0.037

Table 4.6. Parameters of woody plant cover structure (trees, shrubs and vines structural information) compared between nesting (used) and general (available) habitats of Black-throated finch. (Variable codes: DBH= diameter at breast height; DTS= dead tree standing per hectare; and DTG= dead tree on the ground; Total abundance or total richness = abundance or richness of shrubs and trees within all DBH classes; $\bar{x}$ = mean; SD= standard deviation; SE= standard error; Min= minimum value observed; Max= maximum value observed).

Parameters	**Used**					**Available**					**t value**	**df**	**p-value**
	$\bar{x}$	SD	SE	Min	Max	$\bar{x}$	SD	SE	Min	Max			
Shrub abundance (ha^{-1})	395.9	713.8	159.6	0	2788.5	397.0	450.8	63.7	0	1880.0	0.006	25.297	0.995
Small trees abundance (ha^{-1})	181.5	256.6	57.4	0	999.6	477.2	479.1	67.7	0	1560.0	3.330	62.102	0.001
Medium trees abundance (ha^{-1})	63.1	55.6	12.4	0	210.4	110.2	136.4	19.3	0	800.0	2.051	67.934	0.044
Large trees abundance (ha^{-1})	1.3	5.9	1.3	0	26.3	9.0	14.6	2.1	0	70.0	3.138	67.971	0.003
Total abundance (ha^{-1})	641.9	808.6	180.8	26.3	2946.4	993.6	886.2	125.3	10.0	3720.0	1.599	38.221	0.118
Basal area	3.0	2.8	06	0	11.5	3.6	3.1	0.4	0	15.5	0.787	39.908	0.436
DTS (ha^{-1})	15.8	27.5	6.1	0	78.9	12.6	22.2	3.1	0	110.0	-0.461	29.408	0.648
DTG (ha^{-1})	18.4	30.9	6.9	0	131.5	12.0	21.8	3.1	0	110.0	-0.848	26.949	0.404

Fourteen species of trees were used for comparison between used and available habitat (13 trees with 15 or more individuals per hectare plus *Ziziphus mauritiana*; Appendix C, Table C.2). Of those, four species were significantly different; density of *Corymbia clarksoniana*, *Eucalyptus drepanophylla*, *Eucalyptus platyphylla* and *Planchonia careya* was considerably lower in used versus available habitat (Appendix C, Table C.2).

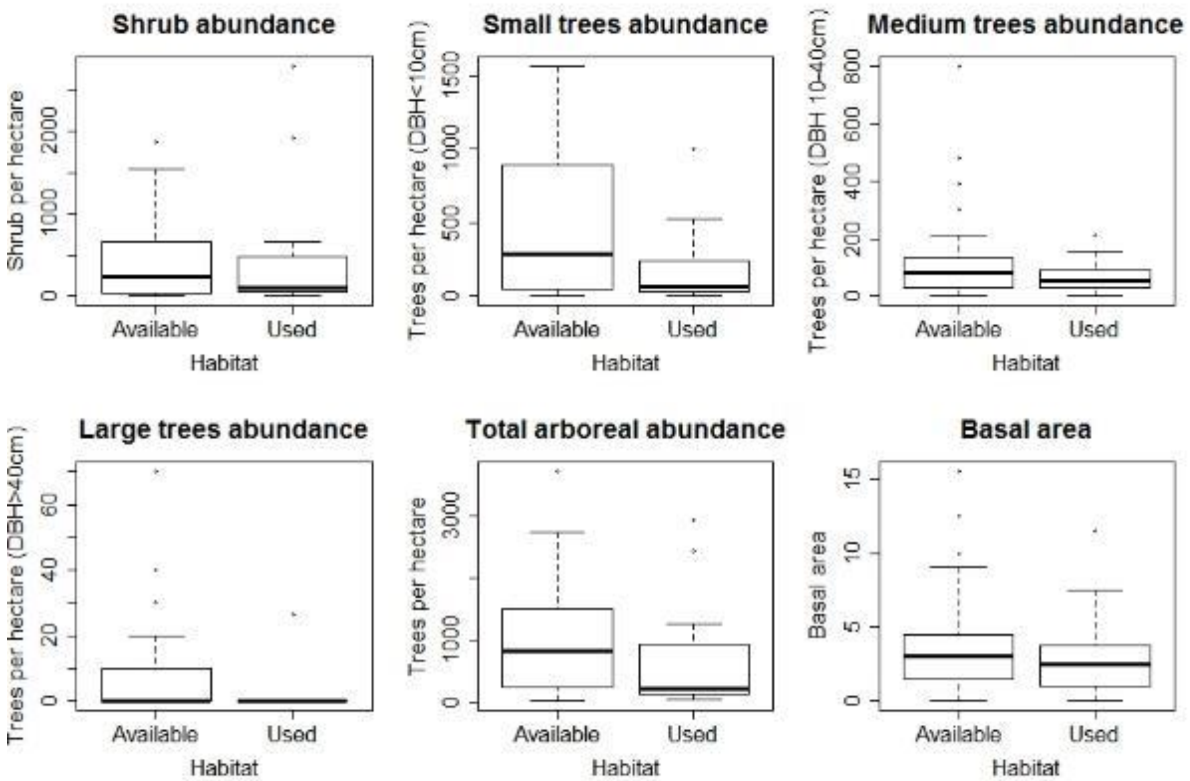

Fig. 4.2. Comparison of woody plant cover parameters between nesting sites (used areas) and general area (available habitat) by BTF.

Nesting sites model

Eleven variables were used to build a model to predict nesting areas for BTF; eight variables were chosen based on the significance of analyses above (1-8) and three were chosen based on field observations and possible relevance for BTF habitat (9-11): (1) Shannon Diversity Index for grasses; abundance of (2) small trees (DBH<10), (3) medium trees (DBH 10-40cm), (4) large trees (DBH>40cm) and (5) Shannon Diversity Index for trees; and abundance of (6) *C. clarksoniana*, (7) *E. platyphylla* and (8) *E. drepanophylla*, and percentages of (9) *Heteropogon contortus*, (10) *Stylosanthes scabra*, and (11) *Themeda triandra* (Appendix C, Table C.2).

In general, nest site selection for BTF was best explained by density of *Eucalyptus drepanophylla* and the Shannon Diversity Index for grasses (present in the five best models). The grass species added in the global model (including the shrub. *S. scabra*) also were present in most of the models. The Shannon Diversity Index for trees/shrubs and abundance of large trees were the only woody plant cover structure variables incorporated into the final models (Table 4.7).

Based on the dredge selection, the best predictive model of BTF nesting habitat showed that BTF are positively correlated with percentage of the grasses *H. contortus*, *S. scabra*, and *T. triandra* and negatively correlated with densities of the canopy tree *E. drepanophylla*, diversity of grasses, diversity of trees/shrubs and abundance of large trees (Table 4.8). The last variable, although not significant, was consistent with our findings on nest site selection.

Table 4.7. Model ranking for nest site selection of the Black-throated finch southern subspecies (*Poephila cincta cincta*) using logistic regression; the function dredge in the MuMIn package (Bartoń 2015) was used to test all possible combinations of the selected variables. Also shown are the ΔAIC (difference in AIC value in regard to best model) and AIC weight. Models are ranked according to the ΔAIC.

Model	Model formula	df	AICc	ΔAICc	Weight
1	*E. drepanophylla* + Shannon Index for grasses + % of *H. contortus* + % of *S. scabra* + % of *T. trianda* + Shannon Index for trees	7	49.41	0.00	0.46
2	*E. drepanophylla* + Shannon Index for grasses + % of *S. scabra* + % of *T. trianda* + Shannon Index for trees	6	51.37	1.96	0.17
3	*E. drepanophylla* + Shannon Index for grasses + % of *S. scabra* + % of *T. trianda* + abundance large trees	6	51.94	2.53	0.13
4	*E. drepanophylla* + Shannon Index for grasses + % of *S. scabra* + % of *T. trianda* + Shannon Index for trees + abundance large trees	7	51.96	2.55	0.13
5	*E. drepanophylla* + Shannon Index for grasses + % of *H. contortus* + % of *S. scabra* + % of *T. trianda* + abundance large trees	7	52.19	2.87	0.11

AIC, Aikake Information Criterion.

Table 4.8. Final model with most relevant variables to predict nesting habitat for BTF in Townsville.

	Estimate	Std. Error	z value	p
(Intercept)	3.2757	1.4161	2.313	0.0207
***E. drepanophylla* abundance**	-1.0152	96.8750	-0.010	0.9916
Grasses Shannon Index	-4.2031	1.3718	-3.064	0.00218
% *Heteropogon cortortus*	0.2009	0.0991	2.027	0.04267
% *Stylosanthes scabra*	0.2695	0.1266	2.130	0.03321
% *Themeda triandra*	0.3817	0.1663	2.295	0.02175
Tree Shannon Index	-2.3406	0.9849	-2.376	0.01748

Discussion

Nesting tree parameters and nest tree selection

In this study, BTF nested in four tree species, preferentially using *Eucalyptus platyphylla* and *Melaleuca viridiflora* for nesting, consistent with findings of other studies (Zann 1976a, Higgins et al. 2006a). From the 71 nests examined by Zann (1976), 60 (84.5%) were in *Melaleuca nervosa* or *M. stenostacha* (Zann 1976; Higgins 2006). Other granivores also have high selectivity in nesting tree species; Blue waxbills (*Uraeginthus angolensis*) placed 79% of its nests in one species of tree (*Acacia tortilis*) and Melba finch (*Pytilia melba*) more than 60% of nests also in *A. tortilis*. These rates are higher than expected based on the tree's abundance (Barnard and Markus 1990). Across its range*,* BTF inhabit areas with 23 different REs in Central Queensland (Vanderduys et al. 2016) and 17 in north Queensland (BTFRT 2007a). Few of these REs in central Queensland contain *E. platyphylla*; therefore, BTF must be selecting other tree species for nesting. In Townsville Coastal Plain, BTF's preference for *E. platyphylla* and *M. viridiflora* should be considered in land management plans. This is particularly pertinent for planning rehabilitation and also for assessing impact of development. Additionally, retaining sufficient a density of trees of the right species is important.

BTF have been known to build domed nests either in the foliage of trees or in tree hollows (BTFRT 2007a, Garnett et al. 2011a). However, most nests found in this study were in the foliage. Field data indicate that, on the Townsville Coastal Plain there is a very low density of trees large enough to sustain hollows: DBH greater than 40cm were rare (Rechetelo, unpublished data). Number and size of hollows in eucalypt trees are significantly correlated with the tree diameter (Bennett et al. 1994, Lindenmayer et al. 2000, Webb et al. 2012); therefore the shortage of hollow bearing trees may have contributed to the lack of nests in hollows. Further evidence for the likely importance of tree hollows for BTF is that the closely related Long-tailed finches (*Poephila acuticauda*), congener of BTF, breeds in hollows (van Rooij and Griffith 2009). Further studies are needed to ascertain preferences for tree hollows where they are available, and the effect of a lack of available tree hollows on nesting success, since birds that nest in hollows usually have higher nesting success (Best and Stauffer 1980).

In this study area, BTF nests were commonly recorded in the top fourth of the trees, which broadly concurs with other studies which found that from 71 nests observed, most of them were in the upper half of the tree (Zann 1976a). Nest heights recorded in literature dealing with the northern subspecies (*P. c. atropygialis*) varied from 1.5-12.2m (Bourke and Austin 1947, Higgins et al. 2006a). Nest position reported for other Estrildid finches is variable: Melba finches (*Pytilia melba*) would nest below eye level, while BTF, Mangrove finches (*Camarhynchus heliobates*) and Blue waxbills (*Uraeginthus angolensis*) have their nests in the upper parts of trees (Barnard and Markus 1990, Fessl et al. 2011). The placement of the nest high in the tree might be related to capturing heat from the sun for a portion of the day (Kendeigh et al. 1977), avoiding heat stress through insolation (Ricklefs and Hainsworth 1969) or avoiding predation (Djomo Nana et al. 2014). The positive relationship between nest height and tree height might indicate that they are not selecting a specific height but proximity to leaves, perhaps to regulate temperature or decrease detection by predators.

BTF rarely nested in mistletoe in this study. Some finch species, such as diamond firetails *Stagnopleura guttata*, preferentially use mistletoe for nest sites (Cooney and Watson 2005). Nests of most granivorous birds are placed in sheltered situations where they are protected by overhanging vegetation; again for thermoregulation or avoiding predation. Protection may be provided by the thick mistletoe foliage. Another factor is that mistletoes are more abundant near edges and fragmented areas with an intermediate level of cover than they are in the middle of patches or continuous landscapes (Turner 1991, MacRaild et al.

2010). In this study most of the nests in mistletoe (eight out of nine) were found at only two sites that were accessed by cattle, have high weed densities (ATC see Chapter 2 for reference) or were used for other activities. On the other hand, trees must be a certain size and age to host a mistletoe plant large enough to support a firetail nest (Cooney and Watson 2005) so the absence of large trees (DBH>40cm) in the study area might be influencing the availability of mistletoe.

The majority of nests found in our study were within 400m of the water source. These data corroborate the information in the management guidelines for BTF, which suggests an ideal nesting site is within 400m of water, because BTF need to drink every day (BTFRT/NRA 2011, Buosi 2011). It was also noted for Gouldian finches (*Erythrura gouldiae*) in northern Australia that suitable nesting habitats should include water sources (Tidemann et al. 1992). However, birds using the nest that was more than 1000m from water undertook daily trips to the water source, and formed large flocks around the same food and water source, even though there were other nearby water sources (Chapter 2).

The majority of the BTF nests were less than 100m from the nearest nest, forming loose colonies. This corroborates previous studies on the BTF (Buosi 2011), and is similar to the nesting behavior of other Estrildid finches, such as Zebra finches, *Taeniopygia guttata* (Zann 1976a, Mariette and Griffith 2012). The formation of a colony can be associated with limited resources such as food, water or nesting substrate, decreasing predation either by diluting predation pressure or by improvement of defense of nests by all nesting birds; or with the conspecific attraction hypothesis, where the presence of a nest is used as a cue for patch suitability, assuming nests are not randomly distributed regarding patch quality (Picman et al. 1988, Picman et al. 2002, Safran 2004, Mariette and Griffith 2012). Little is known about nesting density of BTF and further studies should investigate the relationship of nesting density with the quality of the nesting habitat and breeding success.

Nest site selection

BTF select nest sites with low tree density. Therefore, extensive areas of woody thickening could be a disadvantage to BTF, as it has been shown to be for other granivorous birds such as the Golden-shouldered parrot (*Psephotus chrysopterygius*) and ground foragers in Africa (Garnett and Crowley 2002, Seymour and Dean 2010) among others. Availability of suitable nesting sites with low tree density may drive nesting habitat selection, as has been shown in Gouldian finches (Brazill-Boast et al. 2011). These findings support the BTF management guidelines that highlight the importance of woody vegetation control and maintaining open woodland habitats (Buosi 2011). Land use changes will have effects on vegetation structure; the infrequency of fire, usually in pastoral areas in Queensland, will lead to woody thickening (Russell-Smith et al. 2001). An increase in woody plant abundance induces a reduction in herbaceous production and shifts in composition (Archer 1990, Moreira 2000). Depending on the particular ecosystem, grazing can promote woody thickening by reducing competition from perennial grasses and spreading seeds of woody plants (Knoop and Walker 1985, Brown and Archer 1988, Harrington 1991) or by reducing fuel loads and hence fire intensity (Archer 1995, Roques et al. 2001, Van Langevelde et al. 2003, Beringer et al. 2007). Continuous, heavy grazing will reduce the ground biomass, leading to less intense fires, reducing damage to trees and hence increasing woody establishment (Roques et al. 2001, Van Langevelde et al. 2003).

Results found no selection for BTF nesting sites based on ground cover parameters. The structural variables in ground cover (visual obstruction, vegetation density, percentage of bare ground or rock among others) around nesting habitats did not differ significantly from available habitat. For Sage Sparrows (*Amphispiza belli*), the amounts of bare ground and litter were significantly lower at nest sites than in the overall habitat (Willey 1997). BTF feed on the ground on the fallen seeds of grasses and herbs (BTFRT/NRA 2011) so ground cover structure is expected to have some effect on their foraging areas, but this result points to the importance of other factors not relevant to foraging when selecting nest sites. Nesting sites could be influenced by ground cover and structure in the landscape surrounding the study location. The sampling methodology used in this study measured only variables that were directly adjacent to the nesting sites. Unexplained variation could be because interacting variables in a larger scale than measured, relevant variables were not measured at varying spatial and temporal scales or because the key criteria in the immediate vicinity of nest sites relate to the tree and shrub strata, not the ground layer.

BTF showed a preference for sites with lower tree and grass diversity. Other environmental features in the nesting habitat might be more important to the birds, particularly structural features. The grass species *Themeda triandra, Heteropogon contortus* and shrub *Stylosanthes scabra* were found to have a positive relationship with nesting areas in the predictive model. However, these species seem to negatively impact the suitability of a site for BTF foraging, as per field observations and assessments in the BTF management guidelines. Field observations show that *S. scabra* changes the vegetation structure considerably and the birds cannot access the ground. *T. triandra*, on the other hand, was the most common species at a site with a high probability of occurrence of BTF on the Townsville Coastal Plain and its ecological importance to BTF should be investigated. The positive relationship of all three species in our model could be a consequence of the low number of nests in our sample or because they are widespread in the area and the model showed a different association. Besides, different parameters might be important for different activities of a species. BTF seems to nest in areas with these three species but forage elsewhere. Locations with suitable seeds and grass species are important for foraging activities (details in Chapter 4) (Buosi 2011) but less for nesting. BTF will select nesting areas differently from foraging areas, having specific requirements for each. It is also possible that foraging further from the nest helps reduce the chance that predators will be attracted to the nest.

The idea that BTF nest in the same general location every year (Isles 2007) was not corroborated in our study. Study sites were monitored for almost three years and, although birds sometimes used the same area (within hundreds of metres), nesting location (within dozens of metres) would change from one year to the next. Changing nesting and roosting sites is a strategy for predator and parasite avoidance in other species.

Although no seasonal differences in ground and woody plant cover parameters were tested in this study, variation in choice of nesting habitat may occur between seasons. The breeding success of some savanna birds is more influenced by season than by vegetation and landscape features (Galligan et al. 2006). There was no selection by BTF for nesting locations with particular ground cover parameters but they showed a preference for locations with low tree abundance. Ground cover is more sensitive to changes, such as rain (wet and dry seasons) and fire, and woody plant cover would be more stable. Although time of sampling should be consisdered in further studies, basic structural features of woody cover found in this study are likely to be kept from one season to the other.

This was the most substantial research on BTF nest site selection to date. Despite the moderate sample size in our study, our findings highlighted important features of nesting areas; particularly the lower density of trees in nesting areas, including the preference for specific trees for nesting (*E. platyphylla* and *Melaleuca* spp, for Townsville Coastal Plain). There will be differences across the range due to different flora and conditions (e.g. BTF might prefer different tree species for nesting in Central Queensland). Therefore, this study helps to understand the requirements of BTF on Townsville Coastal Plain, but might as well help in different areas as the structure (ground cover and woody plant density) is likely to be similar. From a management perspective, BTF habitat should be managed in ways that discourage woody thickening. This study also corroborates important information for BTF conservation: the proximity of water sources in nesting habitat. BTF nesting requirements should be included in management actions for the species.

Chapter 5

Selection of foraging patches by the Black-throated finch southern subspecies (*Poephila cincta cincta*) in north eastern Australia[4]

Abstract

The structure of herbaceous vegetation can affect food availability and accessibility to granivorous birds, which impacts on the suitability of habitat for foraging. This study aimed to identify the important features of the vegetation at foraging sites used by the Black-throated finch southern subspecies *Poephila cincta cincta*, an endemic and endangered granivorous bird from eastern Australia, to improve understanding of foraging preferences as a basis for conservation management. From 2011 to 2014, vegetation structure and composition were quantified in patches used for foraging by Black-throated finch (BTF), in neighboring areas and generally available habitat. A total of 33 foraging patches were surveyed. Vegetation density was significantly lower in used areas than in neighbouring and available areas for all strata measured. Used areas had significantly less visual obstruction than neighbouring and available areas, had more bare ground, and significantly lower vegetation cover. Shrub density was also significantly lower in used areas. Selection of foraging patches by BTF was best explained by the presence of dead trees and lower visual obstruction. Although foraging patches were less densely vegetated than the generally available habitat, they adjoined areas with high grass structural complexity (higher visual obstruction, higher vegetation density in almost all strata measured, higher vegetation cover, particularly grass cover, and more species per ha). This indicates that foraging patches consist of open areas (little grass cover) close to grassy areas. Dead trees may play an important role in BTF foraging patches as an intermediate stratum to access the ground. A landscape mosaic containing areas with a high density of suitable grasses as well as open patches would provide sufficient food that was highly accessible to the birds. Control of shrub proliferation, but not complete clearing of the midstorey so leaving features that can be used for resting and shelter, may also enhance this mosaic.

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[4] This chapter will be submitted as a paper:
Rechetelo, J., Grice, A.C., Reside, A.E., Hardesty, B.D. and Moloney, J. 2015. Selection of foraging patches by an endangered granivore, the Black-throated finch southern subspecies (*Poephila cincta cincta*) in north eastern Australia.

Introduction

The abundance, availability and distribution of food strongly influence the population dynamics of animals (Brandt and Cresswell 2008), by influencing the choice of foraging locations, which elements of the environment are exploited, duration of foraging, and which of the available food(s) are consumed (Hayslette and Mirarchi 2002, Jones et al. 2006, Milesi and Marone 2015). Specific patches where animals are exploiting resources differ in nature and appearance from the matrix in which they are embedded (Rotenberry and Wiens 1998). Individuals will be attracted to, and preferentially use, the best available habitat patches (Fretwell and Lucas 1969) given that resources are non-uniformly distributed in space (Graham and Blake 2001). How birds use the landscape, and so their occurrence and persistence within the landscape, will be affected by composition and structure not only of the foraging patch, but also by how habitat patches are arranged with respect to one another (Graham and Blake 2001, Butler et al. 2005b, Jones et al. 2006).

Selection of foraging patches can involve a combination of microclimate and microhabitat, whereby the animal optimizes food searching while minimizing risks (Villén-Pérez et al. 2013). In Australian savannas, several aspects of vegetation structure affect the foraging preferences and patch use by animal populations (Kutt and Martin 2010, Price et al. 2010, Kutt et al. In Press). Changes in land use and its effects on the local fauna in Australian savannas have been extensively discussed (Woinarski 1990, Woinarski and Recher 1997, Woinarski et al. 1999, Woinarski and Ash 2002, Whitehead et al. 2005, Kutt and Woinarski 2007, Woinarski and Legge 2013), though the specific fine scale requirements – i.e. vegetation structure and composition – of many species are still unknown.

Savanna granivore assemblages are greatly affected by the abundance, availability and accessibility of seeds, and in Australian tropical savannas, the abundance of seeds on the soil surface varies seasonally. The greatest concentrations occur at the end of the wet season after which there is a gradual decline until the start of the wet season (Franklin et al. 2005, Williams et al. 2005, Woinarski et al. 2005). Rainfall will trigger germination, leading to food shortages at the onset of wet season (Crowley and Garnett 1999). Granivorous birds of the tropical savannas move through the landscape looking for resources that are heterogeneously distributed (Zann et al. 1995, Clarke 1997, Dean 1997, Brandt and Cresswell 2008). Vegetation architecture directly and indirectly influence the abundance and type of resources available, providing clues to habitat suitability (Deppe and Rotenberry 2008). Understanding how herbaceous vegetation structure affects food availability and accessibility to granivorous birds will help devise effective management practices.

Granivorous birds of northern Australian savannas have declined substantially since European settlement, and species within this assemblage that forage on the ground are thought to be most affected (Franklin et al. 2005). The Black-throated finch, a ground-foraging granivore (Buosi 2011), is an endemic species of north eastern Australia. Its southern subspecies (*Poephila cincta cincta*) has experienced significant declines and is now listed as endangered under Commonwealth and State legislation (New South Wales Government 1995b, EPBC 1999, Queensland Government 2006a). This study aimed to identify the important features of the vegetation at foraging sites used by the Black-throated finch southern subspecies (herein referred to as BTF), and to improve understanding of foraging preferences as a basis for conservation actions and management practices.

Methods

Study area

Vegetation surveys were carried out in the vicinity of Lake Ross, south of Townsville, Queensland, north eastern Australia during the dry season from June 2013 to January 2014. Eucalypt open-woodlands are the dominant vegetation type (Sattler and Williams, 1999, Murtha, 1975). The ground vegetation layer is generally dominated by several native grasses but introduced species – grasses, shrubs and trees – dominate the ground layer in many areas (detailed description in Chapter 1) (Murtha 1975, Smith 2002, Grice et al. 2013).

Data collection

Patch use by BTF in this study indicates their preferred foraging areas within a matrix of general habitat. The specific patches (<10 m^2) where birds were observed foraging were located by searches of the study area and by following radio-tracked individuals (Chapter 2). Birds were observed until the end of the foraging event, and once they moved away from the patch, it was marked using a global positioning system (Model GPSmap62s, Garmin). A total of 33 foraging patches were surveyed. The distance of the foraging patch from the closest water source was estimated using GIS; distance to water was subsequently categorized: <400 m, 400-1000 m and >1000 m), as in Chapter 4. In this study, foraging patches where birds were sighted are defined as 'used areas', areas adjacent to the foraging patch, but not used by the birds, are called 'neighboring areas', and the overall environment where BTF flocks were commonly seen is called 'available habitat' (Fig 5.1).

Ground cover parameters (structure and composition of grasses, forbs and sedges; percentage of bare ground, litter and cover; vegetation density and visual obstruction (Table 5.1) were assessed within the foraging patch, in a neighboring patch and across available habitat. Ground cover parameters were measured in three randomly placed 0.25 m^2 quadrats in patches where BTF were foraging, three in the neighboring area where they were not foraging, and on 25 x 1 m^2 quadrats along 5 transects in 10 areas in available habitat (detailed description in Chapter 3). Foraging patches were sometimes less than 2m^2 so requiring smaller quadrats to survey ground cover.

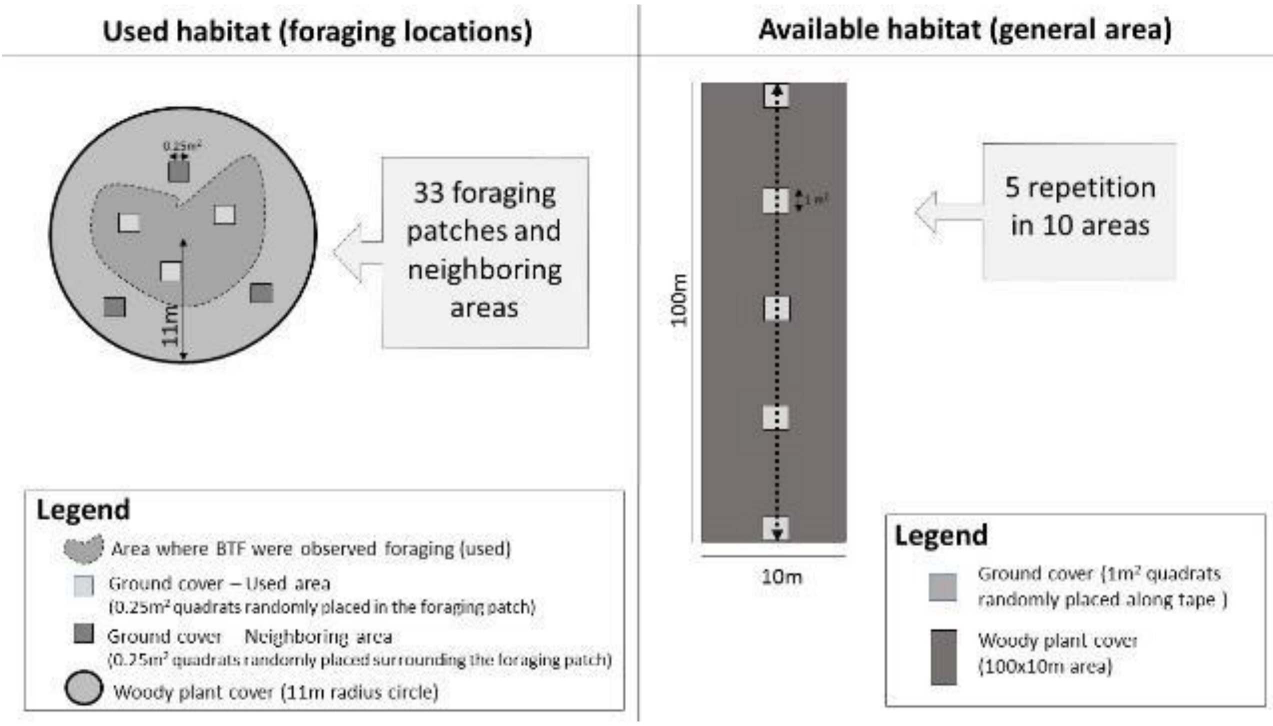

Fig 5.1. Sampling design for vegetation surveys in used, neighbouring and available habitat.

Woody plant cover parameters (structural and compositional measures of trees, shrubs and vines; Table 5.2) were estimated in a circle of 11 m radius around the center of the foraging patch, with all plants within the circle recorded (Martin 1998). Woody plant cover parameters were also assessed within available habitat (10x100 m areas randomly placed across the study sites; detailed description in Chapter 1; Fig. 5.1). *Stylosanthes scabra* is a low growing shrub that was very abundant at some sites and dominated the ground layer. It was counted as part of the ground stratum.

Table 5.1. Ground cover variables measured for BTF used, neighbouring and available habitat.

Variables			**Definition**
Ground cover structure	GCD		Ground cover (vegetation) density: *n* leaves/stems intercepted at a vertical point recorded in 20 cm height increments from ground level. Six classes were analysed based on the stratum of the vegetation: GCD 0-20 cm, GCD 20-40, GCD 40-60, GCD 60-80, GCD 80-100 and GCD >100 cm. Mean was calculated for each class.
	VO		Visual obstruction – a 1m pole was placed in the centre of the quadrat and an observer estimated the visibility from an eye height of 1.0m and 4m from the pole (mean of three repetitions). VO measured in relation to an area with no ground cover, whereby 100% of the pole would be observed.
	FWM		Fallen woody material – *n* of logs larger than 10 cm diameter and 0.5 m long in an area 50 x 10 m along the tape measure
	Total 100%	BG	% of bare ground in a 1m^2 quadrat (BG, rock, litter and ground cover added 100%) estimated in all 5 quadrats along the transect
		Rock	% of rock (as above)
		Litter	% of litter (as above)
		VC	% of vegetation – grasses, forbs, seedlings (as above)
	G_NSpp		*n* grass/forb species within that quadrat
Ground cover composition			Grasses, forbs and sedges identified to species where possible, otherwise genus, or family. Life form and status (native, introduced) recorded. The most visually abundant species within the patch as a whole was recorded.

Table 5.2. Woody plant cover variables measured for BTF used and available habitat

Variables		Definition
Woody plant cover structure	Shrub abundance	*n* individual shrubs per hectare (calculated for the11m radius around nesting trees and in a 100x10 m^2 area along transects). Counts converted in hectare.
	Shrub richness	*n* shrubs species (as above)
	Small tree abundance	*n* individual trees DBH < 10 cm / ha
	Small tree richness	*n* tree species DBH <10 cm / ha
	Medium tree abundance	*n* individual trees DBH 10-40 cm /ha
	Medium tree richness	*n* tree species DBH 10-40 cm /ha
	Large tree abundance	*n* individual trees DBH > 40 cm /ha
	Large tree richness	*n* tree species DBH > 40 cm /ha
	Total arboreal abundance	Total *n* trees and shrubs individuals /ha
	Total arboreal richness	Total *n* tree and shrub species/ ha
	Basal area	Basal area measured in the centre of each transect (50m mark) by using cruise master prism
	DTS	*n* dead trees still standing and used as perch
	DTG	*n* dead trees on the ground
Woody plant cover composition		Trees, shrubs and vines were identified to species where possible, otherwise genus

Data analysis

Summary statistics were calculated for all the vegetation parameters and the Shannon Diversity Index was calculated for ground cover composition (used, neighboring and available areas) and woody plant cover composition (used and available areas). To test if BTF were being selective in their choice of foraging patches, I used Welch's two sample t-test to compare variables (ground cover and woody plant cover) in used and available habitats to identify any differences between foraging patches and the matrix within which they were embedded. I compared the differences in habitat variables across sites (used, neighboring and available) using ANOVA. Shapiro Wilk Normality test and Bartlett's test for Homogeneity of Variances were used to test ANOVA assumptions. Tukey HSD was used for post hoc tests. If assumptions were not met, I used Kruskal-Wallis tests. Density was calculated for tree or shrub species in which there were 15 or more individuals. I compared the density of trees and shrubs at used versus available sites using Welch's t-test.

To investigate the vegetation parameters that were most influencing foraging patch selection I used logistic regression. The variables chosen, as in Chapter 4, were based on both the results of Welch's t-tests and anecdotal ecological observations of BTF, such as grass diversity and bare ground (BTFRT 2007a, Buosi 2011); a group of parameters from the ground cover and woody plant cover was chosen and a global model was fitted. The logistic regression (fitted with the R function gIm; family binomial; binary response) was used to distinguish used from available habitats; the relative importance of the variables was assessed by fitting all possible combinations of the selected variables using the function dredge in the MuMIn package (Barton and Barton 2014, Bartoń 2015) to find the best predictive model for foraging sites. Akaike Information Criterion (AIC) is a method for selecting a model from a set of models (Burnham and Anderson 2002). In this study, AICc was used instead of AIC, as this model corrects for the fact that the number of included parameters was high in relation to sample size (Burnham and Anderson 2002, Barton and Barton 2014, Bartoń 2015). Model selection was performed by using the model with the highest weighted AICc value. Akaike weights give the probability that a model is the best model, given the data and the set of candidate models (Burnham and Anderson 2002). Two foraging patches were excluded from the data due to missing values in vegetation sampling. Variables of ground cover and woody plant cover from used and available areas were used for logistic regression analyses (neighboring area parameters not used). All analyses were performed with the statistical package R (R Core Team 2014) and the packages ‘lattice’ (Sarkar 2015) and ‘car’ (Fox and Weisberg 2011).

Results

Foraging patch selection

Of the ground cover structural characteristics, nearly all variables were significantly different between used, neighbouring and available areas (Table 5.3). Ground cover (vegetation) density was significantly lower in used areas than neighbouring and available areas for all strata measured (GCD 0-20 cm, GCD 20-40, GCD 40-60, GCD 60-80 and GCD 80-100) but the last one (GCD >100 cm; Table 2; Fig. 5.2). Vegetation of neighbouring areas was denser than that of available areas for the second vegetation stratum (GCD 20-40 cm; $p < 0.00$). Although the first and third vegetation strata (GCD 0-20 and GCD 40-60) were not significantly different, they were also denser in neighbouring than available areas (Fig. 5.2).

Table 5.3. Ground cover variables measured in used, neighbouring and available areas (general habitat) of BTF (Codes: GCD = ground cover – vegetation - density in all six classes; VO= visual obstruction; BG= bare ground; VC = % of vegetation cover, separated in Forb% and GS% (Grasses and Sedges%).

Parameters	**Used**					**Neighboring**					**Available**						**p-value**
	$\bar{x}$	SD	SE	Min	Max	$\bar{x}$	SD	SE	Min	Max	$\bar{x}$	SD	SE	Min	Max		
GCD 0-20	3.29	2.33	0.42	0.3	9.7	8.89	5.02	0.90	0.33	23.3	6.98	5.37	0.76	0.2	25.8	$F_{2,109}$= 11.868	< 0.001
GCD 20-40	0.26	0.44	0.07	0	2.0	4.55	3.49	0.63	0	13.3	2.79	2.66	0.38	0	9.8	$F_{2,109}$= 21.906	< 0.001
GCD 40-60	0.07	0.21	0.03	0	9.2	2.49	2.75	0.49	0	8.33	1.78	2.04	0.29	0	9.2	$F_{2,109}$= 12.385	< 0.001
GCD 60-80	0.03	0.09	0.01	0	0.3	1.17	1.50	0.27	0	6.3	0.97	1.29	0.18	0	5.8	$F_{2,109}$= 8.772	< 0.001
GCD 80-100	0.02	0.08	0.01	0	0.3	0.54	1.07	0.18	0	4.7	0.39	0.59	0.08	0	2.4	$F_{2,109}$= 4.979	0.008
GCD >100	0.00	0.00	0.00	0	0	0.15	0.54	0.09	0	3.0	1.71	7.51	7.50	0	44.6	$F_{2,109}$= 1.466	0.235
VO	83.68	14.69	2.64	51.1	100	69.79	21.69	3.89	26.7	100	70.02	22.19	3.14	19.8	100	$F_{2,109}$= 5.165	0.007
BG (%)	40.59	19.28	3.46	2.3	85.0	26.23	17.08	3.07	1.7	56.7	24.40	17.60	2.49	0.4	78.0	$\chi^2(2)$= 15.053	< 0.001
Rock (%)	0.14	0.56	0.09	0	2.7	0.00	0.00	0.0	0	0	0.88	2.12	0.29	0	13.0	$F_{2,109}$= 4.347	0.015
Litter (%)	42.76	18.38	3.30	11.7	76.7	37.38	20.89	3.75	9.0	89.3	45.81	16.39	2.32	16.0	85.4	$\chi^2(2)$= 5.432	0.066
VC (%)	16.51	15.26	2.74	0	49.7	36.39	19.65	3.53	3.3	83.3	28.91	16.46	2.33	1.8	65.0	$\chi^2(2)$= 18.413	< 0.001
Forb (%)	1.86	3.17	0.55	0	12.7	6.02	7.63	1.33	0	31.7	10.38	9.05	1.28	0	36.4	$F_{2,113}$= 13.368	< 0.001
GS (%)	11.17	12.97	2.23	0	40	27.92	23.71	4.13	0	80	16.29	14.56	2.06	0	58.2	$F_{2,113}$= 8.259	0.001

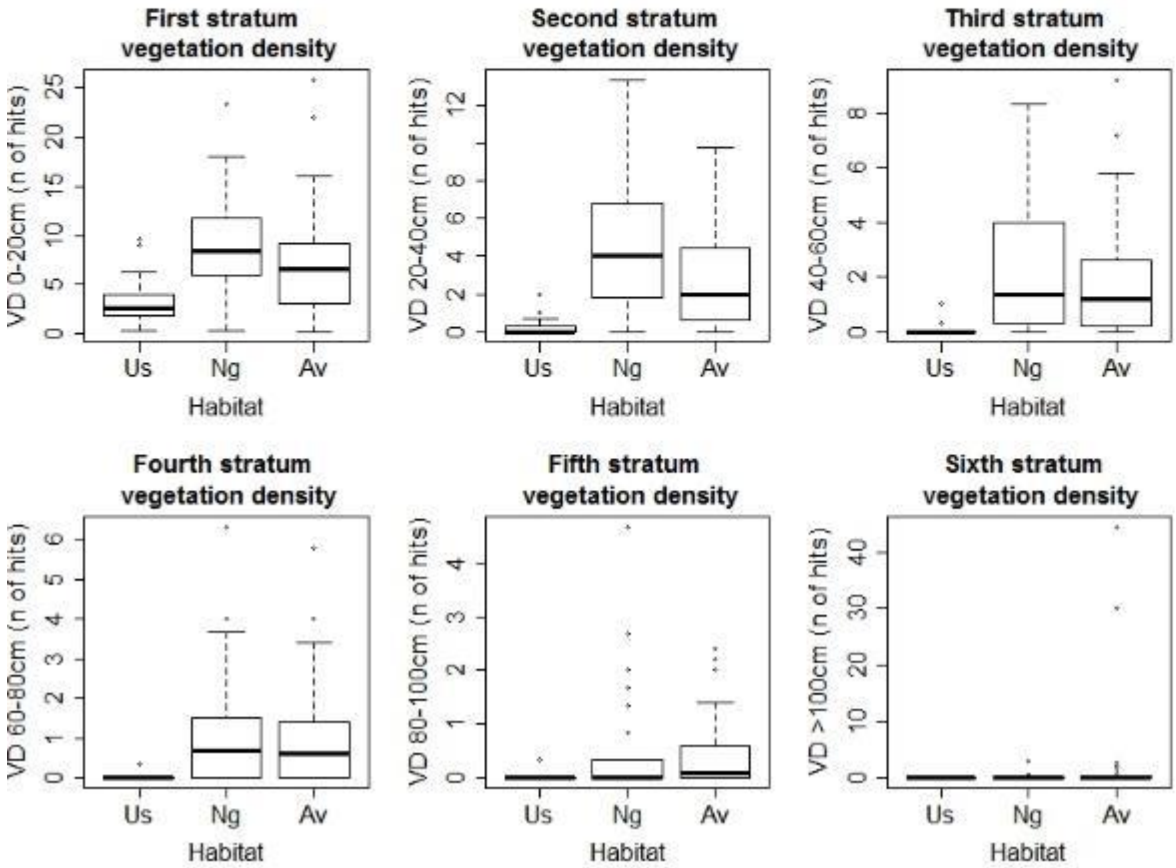

Fig. 5.2. Vegetation density in used, neighbouring and available areas of BTF habitat in different levels (Codes: Us – used habitat; Ng – neighbouring habitat; and Av – available habitat).

Used areas had significantly less visual obstruction than neighbouring and available areas (Table 5.3). Used areas had higher bare ground cover than neighbouring and available areas while neighbouring areas, although not significantly different, had less litter cover (Table 5.3). Used areas had significantly less vegetation cover than neighbouring and available areas, but there was no difference between available and neighbouring areas (Table 5.3; Fig. 5.3). Neighbouring areas had significantly higher percentages of grasses/sedges cover than used or available areas (Fig. 5.3). Used areas had significantly lower percentages of forb cover than neighbouring and available areas; available areas had the higher percentages of forbs in this study (Table 5.3; Fig. 5.3).

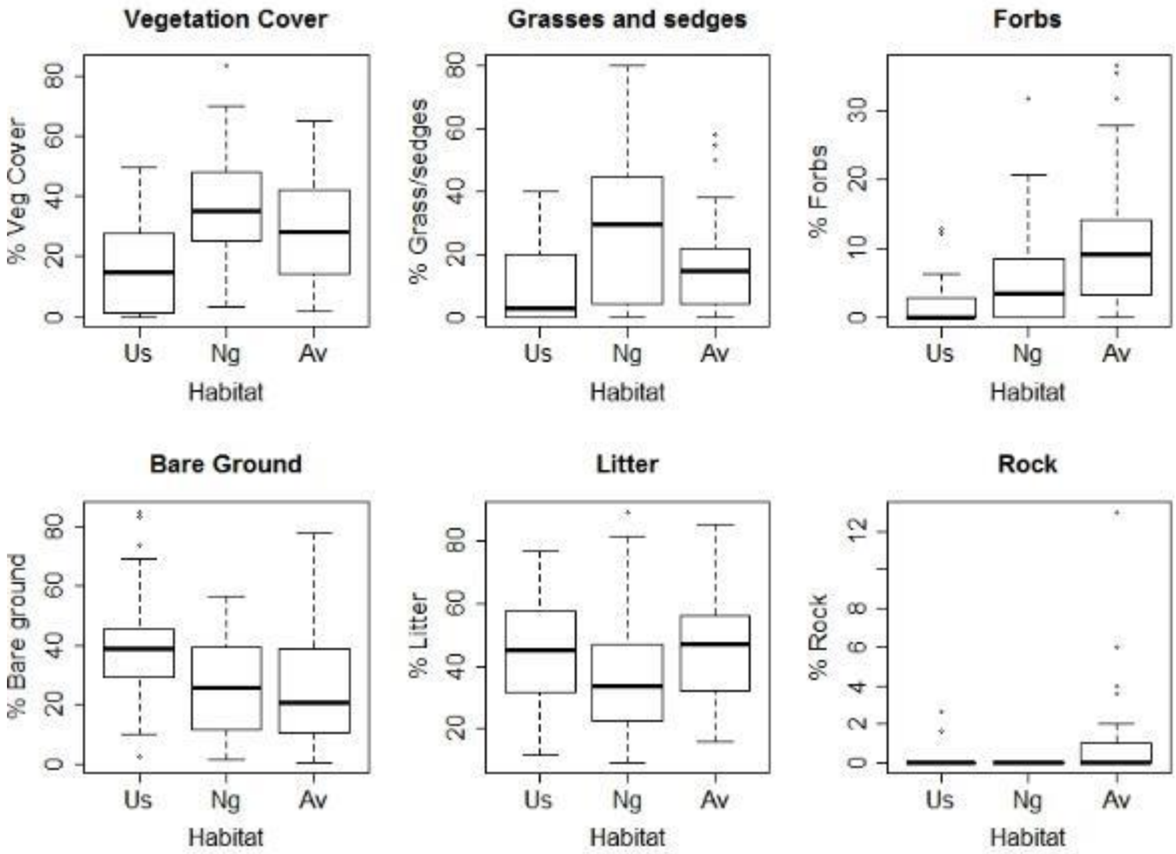

Fig. 5.3. Vegetation cover in used, neighbouring and available areas of BTF habitat. Vegetation cover, bare ground, litter and rock sum up 100%. Vegetation cover was divided in percentage of grass/sedges and percentage of forbs (Codes: Av – available habitat; Ng – neighbouring habitat; and Us – used habitat).

Of ground cover species, 61 species were recorded in all three areas; 56 occurred in available habitat, 30 in neighbouring areas and 27 in used area. Only four species occurred exclusively on used areas (*Bacopa floribunda*, *Galactia tenuiflora*, *Rostellularia adscendens* and *Zornia areolata*; all of them fitted into the classification 'forb'). Available areas had 28 exclusive species. Neighbouring areas did not have exclusive species. The Shannon Diversity Index was significantly smaller in used areas compared with neighbouring and available areas. Of the 15 annual species recorded, only six occurred in used areas (*Ischaemum* spp, *Paspalum scrobiculatum*, *Ludwigia hyssopifolia*, *Portulaca pilosa*, *Hyptis suaveolens* and

Setaria spp.) and seven in neighbouring areas (same species as used areas plus the forb *Alternanthera* sp.). Of the perennial species, 14 out of 29 occurred in used areas and 18 in neighbouring areas. Of species classified as facultative perennial, 6 out of 17 occurred in used areas and neighbouring areas. Thirty-two native species were recorded and of those 12 species occurred in used areas and 10 in neighbouring areas (Appendix D, Table D.1). Twenty species were non-native and of those 12 occurred in used areas and 13 in neighbouring areas (Appendix D, Table D.1). During informal observations made during surveys, BTF were never seen foraging near Shrubby stylo (*Stylosanthes scabra*). One of our study sites had the invasive Grader grass (*Themeda quadrivalvis*) and BTF abundance was negatively correlated with the abundance of this species. While conducting this study, BTF were commonly seeing foraging on the ground and accessing the seed bank; however, in three events BTF were observed feeding on specific items: Red natal grass (*Melinis repens*), Gomphrena weed (*Gomphrena celosioides*) and flying termites (after a rainfall event). Visual estimates of the most common grass species within each patch revealed that *Ischaemum* sp. is abundant in six patches (other species were present in only one or two patches or could not be identified), while neighboring patches had *Ischaemum* sp. and *Themeda triandra* in eight patches each.

Among woody plant cover characteristics, of 10 parameters analysed, used areas had lower shrub density, lower density of large trees (DBH > 40 cm), lower total arboreal density and lower density of FWM (Table 5.4). Although not significantly different, there were more dead trees (standing + on the ground) in used areas (Table 5.4).

Table 5.4. Woody plant cover variables measured in used and available habitat of BTF (Variable codes: DBH= diameter at breast height; FWM= fallen wooden material; DTS= dead tree standing per hectare; and DTG= dead tree on the ground; $\bar{x}$ = mean; SD= standard deviation).

Parameters	Used					Available					t value	df	p-value
	$\bar{x}$	SD	SE	Min	Max	$\bar{x}$	SD	SE	Min	Max			
Abundance shrub (ha)	118.8	186.6	32.5	0	657.7	397	450.8	63.7	10	1880	3.888	70.474	0.000
Abundance tree DBH<10 (ha)	349.2	475.9	82.8	0	1946.7	477.2	479.1	67.7	0	1560	1.196	68.968	0.236
Abundance tree DBH 10-40 (ha)	79.7	63.9	11.1	0	236.8	110.2	136.4	19.3	0	800	1.369	74.43	0.175
Abundance tree DBH>40 (ha)	0.8	4.6	0.79	0	26.3	9	14.6	2.06	0	70	3.705	62.565	0.000
Total arboreal abundance (ha)	548.4	540.6	94.1	0	1946.7	993.6	886.2	125.3	10	3720	2.840	80.598	0.006
Basal area	3.6	2.4	0.41	0.5	10.5	3.6	3.2	0.44	0	15.5	-0.085	79.645	0.933
FWM (ha)	33.5	45.7	7.9	0	184.1	80.4	114.3	16.2	0	540	2.604	69.369	0.012
DTS (ha)	19.9	27.1	4.7	0	105.2	12.6	22.2	3.14	0	110	-1.292	59.002	0.201
DTG (ha)	21.5	26.7	4.6	0	131.5	12.0	21.8	3.1	0	110	-1.707	59.056	0.093
DTS + DTG (ha)	41.5	42.1	7.3	0	184.1	24.6	35.6	5.03	0	150	-1.895	60.475	0.063

Shrub density was significantly lower in used areas and the same was true for density of large trees (DBH > 40cm), total density of trees and fallen woody material (FWM). The variable DTS + DTG was significantly greater in used than in available habitat (Fig. 5.4).

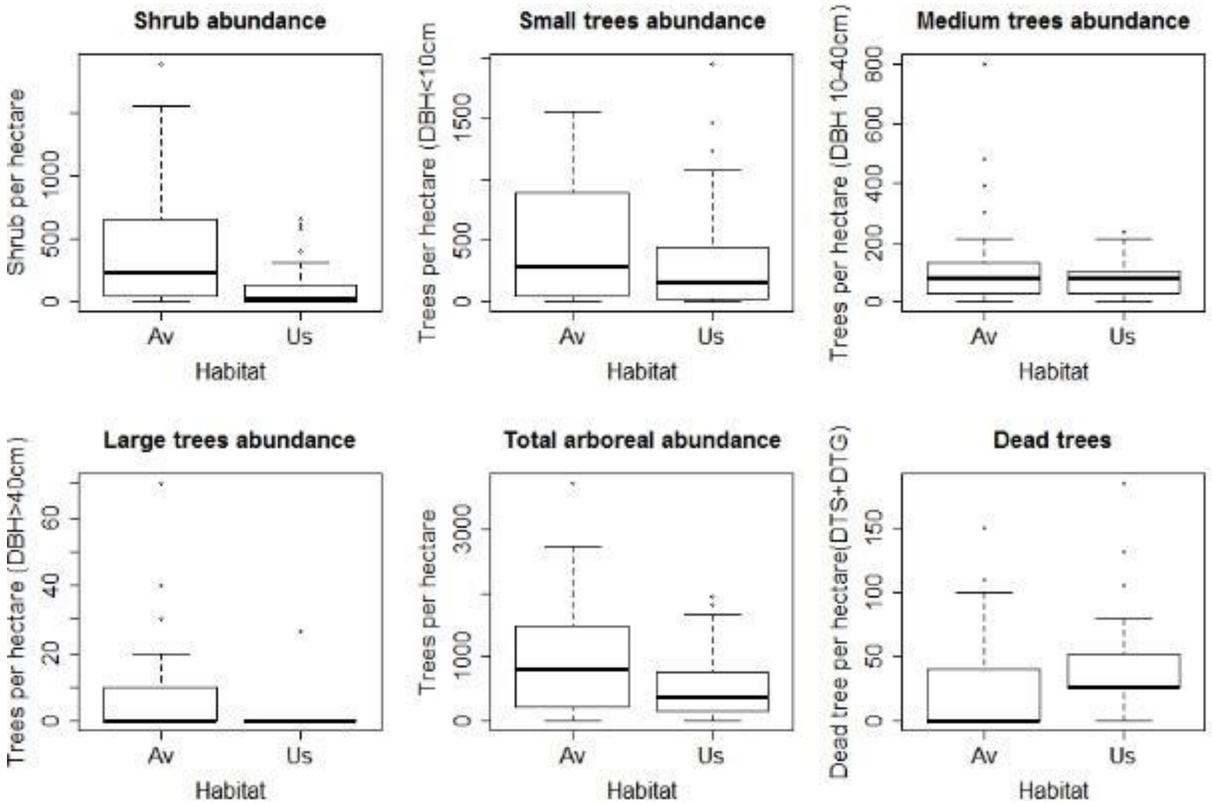

Fig. 5.4. Abundance of trees and shrubs used and available habitat of BTF (shrub, small trees, medium trees and large trees abundances; total arboreal abundance (trees + shrubs) and dead tree (standing and on the ground) abundance).

To examine the effects of woody plant cover on foraging, 14 species were used for comparison between used and available habitat. Of those, five species were significantly less dense in used versus available habitat: *Eucalyptus platyphylla* (t = 2.8062, df = 80.959, p-value = 0.006), *Lantana camara* (t = 2.0064, df = 49, p-value = 0.051), *Lophostemon grandiflorus* (t = 2.1863, df = 49.47, p-value = 0.034), *Planchonia careya* (t = 2.211, df = 50.393, p-value = 0.032) and *Ziziphus mauritiana* (t = 2.7325, df = 50.244, p-value = 0.009)

(Appendix D, Table D.2). The Shannon Diversity Index for trees/shrubs was significantly lower in used areas ($t = 4.7143$, $df = 70.013$, p-value < 0.001).

Foraging habitat model

Based on the results of these analyses and general knowledge of BTF ecology, 11 variables were used to build a global model to predict foraging areas: 1) ground cover density (GCD) 0 – 20 cm, 2) vegetation density 40 – 60 cm, 3) visual obstruction (VO), 4) percentage of bare ground, 5) percentage of grasses/sedges, 6) abundance of large trees (La_ab), 7) shrub abundance (Sh_ab), 8) total tree density, 9) density of dead trees on the ground and standing (DT) per ha, 10) Shannon Diversity Index for trees/shrubs (Sh_TS) and 11) Shannon Diversity Index for grasses/sedges/forbs. The best predictive models had five or six variables (Table 5.5). Overall, selection of foraging patches by BTF was best explained by the presence of dead trees and lower visual obstruction (variables present in all models; Table 5.5). Of the ground cover variables used in the global mode ($n = 6$), percentage of bare ground was the only one that did not appear on the final models. Of the woody plant cover variables used in the global model ($n = 5$), four were incorporated into the final predictive models (dead trees, Shannon Diversity Index, shrub and abundance of large trees).

Table 5.5. Model ranking for foraging patch selection by the BTF using logistic regression; the function dredge in the MuMIn package (Bartoń 2015) was used to test all possible combinations of the selected variables. Also shown are the ΔAIC (difference in AIC value in regard to best model) and AIC weight. Models are ranked according to the ΔAIC.

Model	**Model formula**	**df**	**AICc**	**ΔAICc**	**Weight**
1	DT + % Grass/Sedges + Sha_TS + GCD 40-60 + VO	7	45.86	0.00	0.32
2	DT + % Grass/Sedges + Sh_ab + GCD 40-60 + VO + La_ab	7	46.73	0.87	0.21
3	DT + Sh_ab + GCD 0-20 + GCD 40-60 + VO + La_ab	7	46.91	1.04	0.19
4	DT + Sha_Gr + %Grass/Sedges + Sh_ab + GCD 40-60 + VO	7	47.37	1.50	0.15
5	DT + Sh_ab + GCD 40-60 + VO + La_ab	6	47.66	1.79	0.13

AIC, Aikake Information Criterion.

Based on the dredge selection, the best predictive model of BTF foraging patch showed that foraging patches are positively correlated with 1) dead tree presence and 2) percentage of grasses/sedges; and negatively correlated with grass diversity, tree diversity, vegetation density 40 – 60 cm and visual obstruction (Table 5.6).

Table 5.6. Final model with most relevant variables to predict foraging patches for Black-throated finches southern subspecies (*Poephila cincta cincta*) in Townsville.

	Estimate	Std. Error	z value	p
(Intercept)	13.91470	5.73305	2.427	0.0152
DT_S_G_DEN	0.05637	0.02339	2.410	0.0159
GrShannon	-2.46535	0.96770	-2.548	0.0108
PercGrass	0.10293	0.04645	2.216	0.0267
Tr_Shannon	-2.86129	1.17925	-2.426	0.0153
GCD _040.060	-7.95185	3.19789	-2.487	0.0129
VO_M	-0.12336	0.05807	-2.124	0.0336

Discussion

Observations and analyses in this study clearly indicate that BTF select patches with particular features in which to forage. Most structural vegetation features in foraging patches differed in abundance or density from those of neighboring and available areas, particularly the ground cover features. The vegetation variables (visibility, percentage of grass cover and shrub density among others) that were included in the best models were consistent with some of those described by studies of other granivorous birds, such as sparrows (*Ammodramus savannarum*, *Ammodramus bairdii*, *Ammodramus leconteii*, *Passerculus sandwichensis* and *Pooecetes gramineus*) (Whitmore 1981, Grzybowski 1983), linnets (*Carduelis cannabina*), Grey partridge (*Perdix perdix*), chaffinch (*Fringilla coelebs*), yellowhammer (*Emberiza citronella*), Reed bunting (*Emberiza schoeniclus*) and Corn bunting (*Miliaria calandra*) (Moorcroft et al. 2002).

Foraging patches were less densely vegetated than neighboring and available areas, with consequently less visual obstruction. In areas of dense grass, food items may be inaccessible or more difficult to detect (Whittingham & Markland 2002, Butler & Gillings 2004), predation risk may increase due to the inability of finches to see predators (Metcalfe 1984, Whittingham and Evans 2004, Jones et al. 2006), and foraging efficiency may be reduced because the vegetation constrains the birds' movements (Buchmann et al. 2012). Other granivorous species, such as the Grasshopper sparrow (*Ammodramus savannarum*), similarly selected less vegetated areas (Whitmore 1981). This openness of patches used by BTF might have several advantages for this ground foraging species. Further studies, particularly manipulative studies, are needed to elucidate the behaviors and quantify the processes involved in patch selection.

Although foraging patches were less densely vegetated than the generally available habitat, they adjoined areas with high grass structural complexity (higher visual obstruction, higher vegetation density in almost all levels measured, higher vegetation cover, particularly grass cover, and more species per ha). This indicates foraging patches are open areas close to grassy areas. The model corroborates the positive relationship with the percentage of grasses in used areas. The same pattern was observed for Red-backed shrikes (*Lanius collurio*) which forage at the boundary between long grass and short grass; the long grass serving as a reservoir of large insects that become accessible when they move into the short grass (Whittingham and Evans 2004). Grassy areas that produce a lot of seed are likely to supply seed to adjacent areas where seed supply may be lower but accessibility is higher (because it is more open).

Foraging patches had more bare ground than neighboring or available areas. BTF prefer to feed on seeds fallen on the ground (Buosi 2011), so bare ground near grassy areas increases their access to seeds. A positive relationship between seed availability on bare ground and use by seed eating birds has been observed for granivorous birds worldwide (Whitmore 1981, Grzybowski 1983, Perkins et al. 2000, Moorcroft et al. 2002, Whittingham and Markland 2002) and is considered the most relevant feature in foraging patches for rare farmland birds in Central Europe (Tagmann-Ioset et al. 2012). Bare ground will also increase the detectability of seeds (Whitehead et al. 1995, Whittingham and Markland 2002), the efficiency of predator detection (Grzybowski 1983, Metcalfe 1984) and foraging efficiency by reducing the physiological costs of foraging within dense vegetation (Moorcroft et al. 2002). The presence of bare patches is therefore key to BTF foraging. A landscape consisting of a mosaic of grassy areas with patches of bare ground, important for BTF, has also been found to be favoured by granivorous and grassland birds generally in Europe (Perkins et al. 2000, Whittingham and Markland 2002). It was also observed in this study that the percentage of grasses/sedges was greater in neighboring areas than used or available ones. Additionally, the best model points to a positive relationship with grass/sedge cover. Further studies could focus on the amount of bare ground in dry and wet seasons to try to understand the dynamics of the patches in the landscape and how this dynamism affects foraging behavior.

Dead trees appear to play an important role in BTF foraging patches. Field observations suggest that BTF use them as an intermediate stratum to access the ground. This fine-scale assessment corroborates the findings of Chapter 2, highlighting the relevance of this structural feature. It is necessary to distinguish the presence of dead trees, which are positively related to use, from shrub density which is negatively related to use. High shrub density will affect foraging patches negatively by decreasing the accessibility of the bare ground by the birds, as well as reducing the amount of grass and therefore food abundance (Scholes and Archer 1997, Dohn et al. 2013). Additionally, anecdotal observations indicate that birds might be using dead trees to check for predators.

Another observation made during field surveys is that BTF might preferentially forage in shade. Thermoregulation can be an important factor in foraging patch selection (Villén-Pérez et al. 2013). This behaviour was observed in other Estrildidae (Hamed and Evans 1984) though structural features (e.g. distance to refuges) seem to be more important than temperature (Villén-Pérez et al. 2013). Although I do not address this question here, it is important to acknowledge that factors in addition to vegetation structure and composition might be influencing when and where birds forage.

Despite the moderate sample size in our study, our findings highlighted important features of foraging areas; particularly the preference for areas with low vegetation density, and low shrub density. Vegetation surveys were carried out in the early and late dry season and differences between seasons will not be detected due to the moderate sample size. Ground cover parameters (grass height, density, etc) are sensitive to time of the year, particularly in relation to the rainy season. Therefore, these differences should be acknowledged in further studies. Seed abundance is likely to decline over time though we sampled patches just after foraging events, focusing on the structure necessary for BTF access the seed bank and not on the seed bank itself. Although not assessing seed abundance, the relevance of ground cover structure is apparent. Seed abundance is important but it will not be used if the seeds are inaccessible to BTF because of vegetation structure. Further studies should focus on seed bank and also sampling evenly across different seasons.

BTF are selecting foraging patches with fewer shrubs, as observed in the study of overall habitat requirements (Chapter 3) and nesting site selection (Chapter 4). Increases in the woody vegetation in savannas have been highlighted as an important land change affecting faunal guilds in Australia (Burrows 2002, Kutt and Martin 2010). Although some studies showed that woody thickening might be advantageous to some savanna bird species (Tassicker et al. 2006), others point to negative relationships (Garnett and Crowley 2002).

This chapter clearly shows that BTF are selecting specific conditions in which to forage. Savannas have a marked seasonal cycle and, although vegetation features such as the location of the most suitable patches change throughout the year, this study has identified key features of foraging patches. A vegetation mosaic containing high grass density with open patches would provide a suitable landscape with both food availability and food abundance and accessibility for the birds (Perkins et al. 2000, Benton et al. 2003). Carefully targeted management of habitat structure has the potential to affect foraging patch selection by BTF. Measures could include prescribed fires and controlled grazing. Micro-habitat management that provides not only grasses that are known to be preferentially consumed by BTF but also provides a mosaic of patches – grassy areas interspersed with patches of bare ground – seems to be crucial to habitat management. Based on field observations, bare patches could be as small as $2m^2$ (patches observed while conducting this study varied from 2x1.5m to 17x8m) or as large as $140m^2$, providing they are close to grasses and water sources. Control of shrub proliferation, but not complete clearing of the mid-storey and leaving features that can be used as intermediate strata may enhance foraging suitability in this mosaic.

Chapter 6
General discussion

Species with small, threatened populations are the most susceptible to extinction (Kerr and Currie 1995). Additionally, species with naturally limited ranges are more likely to be threatened than those that are widespread (Pimm et al. 2014). Currently, approximately 13% of the world's bird species are threatened (Pimm et al. 2014). In Australia, 22 bird taxa have become extinct, 13 are critically endangered, 46 endangered and 65 species vulnerable (EPBC 1999). The Black-throated finch southern subspecies (BTF) is classified as endangered at State and Commonwealth levels (EPBC 1999) and its range is estimated to have contracted by 80% in the last forty years (BTFRT 2007a, Buosi 2011). Currently, only two viable populations are known: one on the Townsville Coastal Plain and one in Central Queensland (Chapter 1). Threats to these two populations jeopardize the species' chances of survival.

Understanding important aspects of the species' ecology is critical for their conservation. Knowing the species movement patterns, home range size, nest site selection and foraging patch selection among others will assist with its conservation and recovery. Considering that about 60% of the remaining suitable BTF habitat is under extractive or exploratory mining tenure (Vanderduys et al. 2016), it is vital to understand the bird's requirements as a basis for effective conservation measures.

My work sought to enhance current understanding of the ecology of BTF to assist with its conservation, focusing on home range size, movement patterns, habitat requirements and nest and foraging patch selection. The findings provide information about mobility patterns and highlight the importance of habitat with patches - bare ground surrounded by grassy areas, patches of *Melaleuca* spp., the presence of specific tree species for nesting - within their individual home ranges to meet their ecological requirements. In this chapter the important findings presented in previous chapters will be summarized and synthesized to consider implications for management of the species. Important topics for future research are also proposed.

Summary of major findings

Aim 1: Determine home ranges sizes and movement patterns

Objective 1: Estimate home range sizes and habitat selection of BTF on Townsville Coastal Plain

Home ranges for BTF on Townsville Coastal Plain were small (ranging from 25.5 to 120.9 ha) but increased over time (5-20 days). During the course of this study, few individuals were recorded moving long distances (>15 km); individuals showed a high degree of site fidelity (>600 days at the same site). Home ranges were greater at Site 2, a fragmented area with varied land uses and, in contrast to Site 1, birds moved daily from roosting to feeding areas. BTF at Site 2 may have needed to move further to meet their requirements in the more fragmented landscape. Fragmentation of habitat usually implies habitat loss (Andrén 1994), and species may need to move further to different patches of habitat to meet their ecological requirements. The greater the habitat loss the greater the expected effect on the species, and the more likely it will be that the overall population will decline (Johnson 2001). This is apparent from a study of BTF abundance in sub-divided peri-urban areas compared with rural areas (Whatmough 2010).

Broad habitat selection was analyzed using home ranges of each BTF overlaid with broad vegetation community layers (Regional Ecosystem classification). BTF habitats preferences were not consistent between the surveyed sites. BTF used different habitat at the two sites and throughout the year. Individuals radio-tracked at different times of the year were using different vegetation types. The resources they use are patchily distributed in the landscape and also must be available to them in different periods as well. Additionally, BTF are responding to their environment at a finer scale than vegetation classification analyzed in Chapter 2 (Regional Ecosystems, RE). BTF were using different vegetation types within the home ranges observed in this study; these are not captured in the RE classification. For instance, at both sites where radio-tracking was conducted, BTF were observed using patches of *Melaleuca* spp. to shelter during the hottest part of the day while areas dominated by the introduced shrub *Stylosanthes scabra* were avoided.

Objective 2: Determine local (daily) and larger scale movement patterns of BTF in savannas woodlands

At one site BTF undertake daily movement between foraging and roosting areas. Field observations aligned with tracking data showed that BTF formed small flocks (2-3 individuals) early in the morning, then in mid-morning several small flocks would gather together at the same foraging area (20-40 individuals). Small flocks would disperse in different directions to roost, but would congregate each day and spend most of the day in a much smaller area. These data show an important mobility pattern for the species. At Site 1, although this local daily pattern was not clearly observed, the flocks were also smaller in the early morning and near nests but BTF seem to aggregate later in the morning. While home range data in this study represents the area used by an individual, the area required by a larger flock (BTF population within a site) would be much greater. Therefore, any conservation strategy should take into account not only individual home ranges, but the area used by the population as a whole, taking into consideration home ranges of all individuals. Additionally, radio-tracking BTF provided important of information about their behavior, particularly about detectability. In areas where tracking surveys were carried out and BTF were present, individuals were often neither seen nor heard, although the tracking device proved its presence in the area. This behavior can lead to false BTF ‘absence’ and even populations due to low detectability. This finding has implications for assessment of populations and their conservation management. Therefore, methods such as transects or point counts should be modified to maximize the chances of detecting BTF, and effort should be made to more reliably distinguish between real absences and false absences.

Aim 2. Understand habitat requirements

Objective 3: Identify habitat requirements by examining environmental variables, particularly vegetation structure and composition, important in determining BTF occurrence

BTF flocks were larger in areas with a higher percentage of native grass species, low shrub cover and abundance, high density of dead trees and the presence of some specific grass species such as *Eragrostis* spp. and *Setaria surgens*. The preference for areas with low shrub cover and abundance was also observed at the scale of foraging patches and nesting areas (Chapters 3, 4 and 5). In this study, areas in which the introduced shrubs *Ziziphus mauritiana*, *S. scabra* or *Lantana camara* were abundant were less, or not, used by BTF. BTF forage on the ground, and a shrubby environment negatively affects the availability and accessibility of grass seeds. However, there was a positive relationship with dead trees. The ecological relevance of standing dead trees in this study is not comparable to the relevance of trees to cavity nesting birds, such as woodpeckers in oak savannas (Johnston 2007). In this study, standing dead trees are small finely branched and sparsely distributed in the landscape, apparently used by the birds to access the ground during foraging activities. The dead trees and the shrub *Acacia holosericea* seem to have the same structural function to BTF, as a medium strata which they use when accessing the ground to forage. Although behavioral analysis was not formally undertaken in this study, several field observations showed BTF using those medium strata structures to perch during foraging events, whereby some individuals would stay vigilant in the branches while others foraged on the ground.

Objective 4: Determine features of microhabitat associated with nesting sites of BTF
Objective 5: *Determine importance of the vegetation in foraging patches used by the BTF*

BTF preferably nest in *Eucalyptus platyphylla* and *Melaleuca viridiflora* and 64% of all nests were within 400 m of a water source. Nesting habitat requires lower tree density and lower shrub density than most of what was available in the region. Only one nest out of 50 was in a hollow. In this study area there were few large trees and the availability of hollows is likely to be limited. Ground cover structural parameters do not play an important role in nesting site selection and diversity of trees and grasses were lower in nesting locations. Foraging patches were nearby grassy areas and ground cover parameters (vegetation density, visual obstruction, percentage of bare ground, vegetation cover) were different from available habitat. As with nesting locations, foraging patches require low shrub density. Where all requirements are not met by individual locations, BTF used different areas for each activity.

The nesting and foraging areas selected by BTF have different characteristics with specific requirements. While there were almost no ground cover structural differences between nesting locations and the generally available areas, almost all aspects of ground cover examined differed between foraging patches and generally available areas. Given that nesting and foraging required different characteristics, at Site 2 BTF used separate areas for each activity, travelling on a daily basis from the nesting-roosting areas to the feeding area. Similar behavior has been found in other ground-feeding granivorous, such as Gouldian finches (*Erythrura gouldiae*), which may forage in grassy patches beyond the breeding area (Dostine et al. 2001).

As has been found with Gouldian finches (Dostine et al. 2001), the location of suitable foraging patches for BTF, as well as specific nesting locations (tens of square metres in area), probably vary between years as the landscape mosaic is constantly changing under the influence of rainfall and other factors, including human impacts. Therefore, it is critical that areas maintained and managed for BTF contain suitable nesting areas, suitable roosting areas and suitable foraging areas in proximity to one another to assure BTF can meet all their requirements. The behaviour of BTF in foraging in close proximity to the nesting site during the breeding season (Isles 2007b) should be further investigated. Different features of the larger landscape will be important to BTF for different ecological requirements.

Recognition of the particular features of nesting sites and foraging patches needs to be matched with appropriate management of other parts of the landscape to assure the maintenance of the local population. As suggested for Gouldian finches, it is vital to manage grassy woodlands to create resource patches at adequate scale and density to provide suitable nesting areas as well as appropriate foraging patches to preserve a viable habitat (Dostine et al. 2001). Another relevant feature of the landscape is the overall importance of patches of small *Melaleuca* spp. that BTF used (along with

Acacia holosericea and dead trees) to rest in the shade, and as a mid-stratum from which to access foraging patches and nesting substrate.

Black-throated finch

BTF were observed using different habitats at different times of the day during the brief periods (5-21 days) they were radio-tracked. Early mornings were usually spent in the nesting area doing activities such as nest maintenance; the middle of the day was used for resting, usually in patches of *Melaleuca* spp.; and mid-mornings and afternoons for foraging activities. For different activities, nesting, foraging, resting, they require different specific vegetation structures and compositions within the landscape, and different types of ground and woody plant cover. Over a day, BTF required different habitat arrangements to meet their ecological needs. Comparing how they use different vegetation types (Regional Ecosystems) shows that birds used different vegetation types to track the resources.

Habitat favorable for BTF must provide for all their requirements within a day and across seasons. Features such as: patches with suitable grasses (e.g. *Eragrostis* spp.), patches with bare ground or low vegetation density (ground cover) will allow the birds to access the seed bank provided suitable seeds are also present. The absence of shrubs but the scattered presence of a medium stratum are important. Therefore, large homogeneous areas will not necessarily maintain BTF populations. BTF require a fine-scale mosaic of vegetation within their daily home range: areas with bare ground surrounded by suitable grass species, low shrub density, presence of suitable woody plant cover and the presence of key species (such as *Eucalyptus platyphylla* and *Melaleuca* spp. in the Townsville Coastal Plain).

Future research directions

Although this study has provided a better understanding of movements, home range sizes and ecological requirements of the BTF on the Townsville Coastal Plain, further work is needed to more fully understand how they use the habitat.

In this study 1) home ranges increased incrementally over time; 2) home ranges in fragmented areas were greater than in non-fragmented ones; 3) home ranges were estimated only in the dry season and 4) there was variation in habitat use between locations. Therefore, further investigation should focus on estimating home ranges 1) over longer time periods; 2) in other fragmented and non-fragmented areas; 3) in different seasons and 4) at finer scales than REs in relation to habitat use and habitat selection.

Long distance movements, although rare in this study, should also be investigated. Long term banding and monitoring of water sources would be a first step to get more information about long distance movements. Additionally, it is important to investigate when and where food resources become available, e.g. by investigating grass phenology – this might complement the understanding of long distance movements.

Although foraging and diet studies have been conducted in the past (Mitchell 1996, Isles 2007b), there are still gaps in our knowledge of BTF diet and foraging preferences. In this study BTF foraged nearby specific grass species. It is still relevant to thoroughly understand which grass species are positively and which negatively related to habitat use by BTF, including native and non-native species.

Field observations showed BTF absence from heavily grazed sites (there was little or no grass to produce seeds) and they never foraged in areas with high vegetation density. High vegetation density probably limits access of BTF to the seed bank on the ground and extensive areas with no grass will not provide food resources for the birds.

It was not within the scope of this study to measure the level of grazing or burning intensity or frequency, but it is relevant to understand how those factors affect BTF.

The literature states BTF build domed nests in trees but also use tree hollows (Garnett et al. 2011a). However, in this study only one nest out of 50 was in a hollow. In this study area there were few large trees and the availability of hollows might be limited. It is important to know whether BTF preferentially nest in hollows in areas where they are available. Little is known about nesting density of BTF and further studies should investigate the relationship between nesting density and the quality of the nesting habitat and breeding success.

Understanding which landscape or vegetation features are driving patch selection will also be important for developing and implementing appropriate conservation and management strategies for BTF. Further studies, particularly manipulative studies, would be useful to test different parameters of foraging patch selection.

BTF can be difficult to detect in a single visit to an area, by a simple transect or point count methodologies. Studies on detectability should be undertaken to propose a standardized sampling scheme to survey areas for BTF. These studies will lead to a consistent estimation of presence and abundance.

Implications for conservation

BTF habitat in Townsville Coastal Plain falls within public and private jurisdictions. Management and conservation actions in both jurisdictions are required to maintain the BTF population in the area. Maintaining a mosaic of habitat (key tree species for nesting, patches of low grass density surrounded by suitable foraging grasses, patches of *Melaleuca* spp., control of shrub encroachment and the presence of a water source) within their home range should be considered. This could be achieved through manipulation of habitat structure (Whittingham et al. 2006) through creation of patches of bare ground within suitable grassy areas (Perkins et al. 2000, Whittingham and Markland 2002), shrub and weed control (particularly *Z. mauritiana*, *L. camara* and *S. scabra*) will be important.

This was an intensive study in one small part of the species' range. In different landscapes, vegetation types and climates, behaviour and ecology may differ somewhat. Overlapping all home ranges used by the birds in a fragmented landscape, an area of several hundred hectares would probably be needed in the short-term for a modest population. Nonetheless, to keep a viable population with minimum risk of extinction in long term, the area of suitable habitat needs to be of greater extent. Additionally, it should meet the ecological requirement for the species regarding vegetation structure and species composition.

Concluding remarks

This is the most comprehensive study on the ecology of BTF. It is the first to radio track, estimate home ranges, band and monitor birds and carry out a thorough vegetation survey in available, nesting and foraging areas. These findings have provided a greater understanding of movement, particularly daily movement patterns and highlighted the importance of areas without dense patches of invasive shrubs. Responding to these findings with appropriate management action will contribute to the persistence of the BTF.

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Appendix A

Additional tables and figures from Chapter 2

Table A.1. Resight and recapture table. Sub-set of the most relevant time and distances travelled by Black-throated finches at Townsville coastal plain.

ID	Band-number	Total resights/recaps	Date		Days banded	Distance travelled (m)
			Banded	Resighted /Recaptured		
1	026-92534	4	03/05/2012	22/05/2013	384	107.11
				19/06/2013	412	98.10
				14/11/2013	560	185.10
				04/02/2014	642	16168.64
2	026-92570	5	07/06/2012	10/08/2012	64	5.10
				15/08/2012	69	1075.61
				17/10/2012	132	1075.61
				31/01/2013	238	285.50
				03/02/2013	241	949.70
3	026-92571	2	07/06/2012	13/12/2013	554	267.68
				09/02/2014	612	294.42
4	026-92572	3	07/06/2012	31/01/2013	238	213.21
				31/01/2013	238	188.66
				03/02/2013	241	949.68
5	026-92577	1	07/06/2012	17/10/2012	132	17012.22
6	026-92592	1	18/08/2012	30/08/2013	377	36.70
7	026-92595	1	18/08/2012	22/11/2012	96	6168.18
8	026-92674	1	10/11/2012	26/01/2014	442	53.38
9	026-92684	1	11/11/2012	31/01/2014	446	736.09
10	026-92689	1	11/11/2012	26/01/2014	441	53.38
11	026-92696	2	30/11/2012	20/12/2013	385	42.62
				26/01/2014	422	43.09
12	026-92713	1	09/12/2012	26/01/2014	413	43.09
13	026-92721	1	22/05/2013	10/12/2013	202	5000.63
14	026-92739	5	30/07/2013	17/09/2013	49	16074.52
				15/01/2014	169	15862.15
				20/01/2014	174	16521.54
				31/01/2014	185	16323.92
				31/01/2014	185	16399.78

Fig. A.1. Asymptotes generated for each radio-tracked individual of BTF at Site 1.

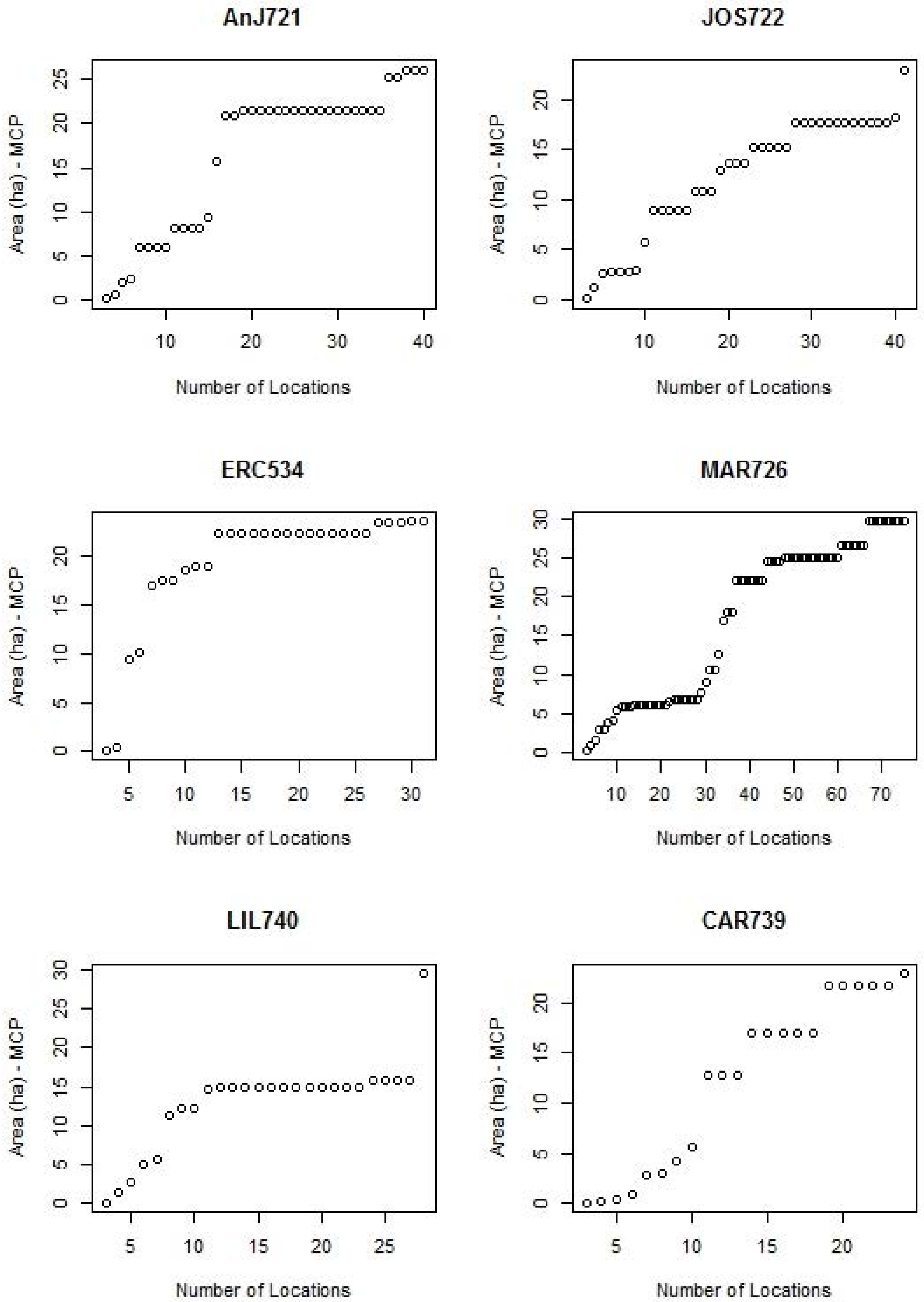

Fig. A.2. Asymptotes generated for each radio-tracked individual of BTF at Site 2.

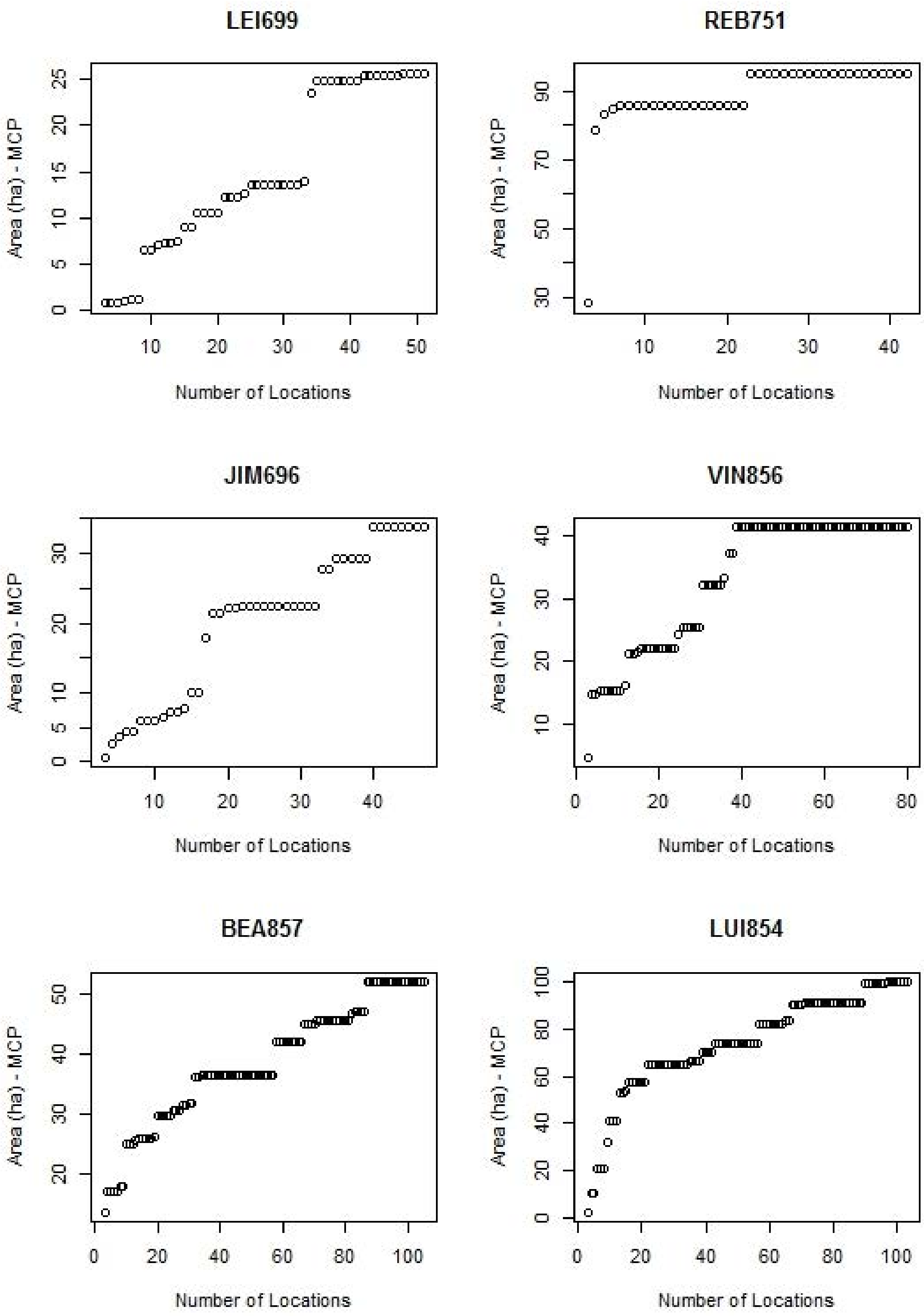

Fig. A.3. Asymptotes generated for each radio-tracked individual of BTF at Site 2.

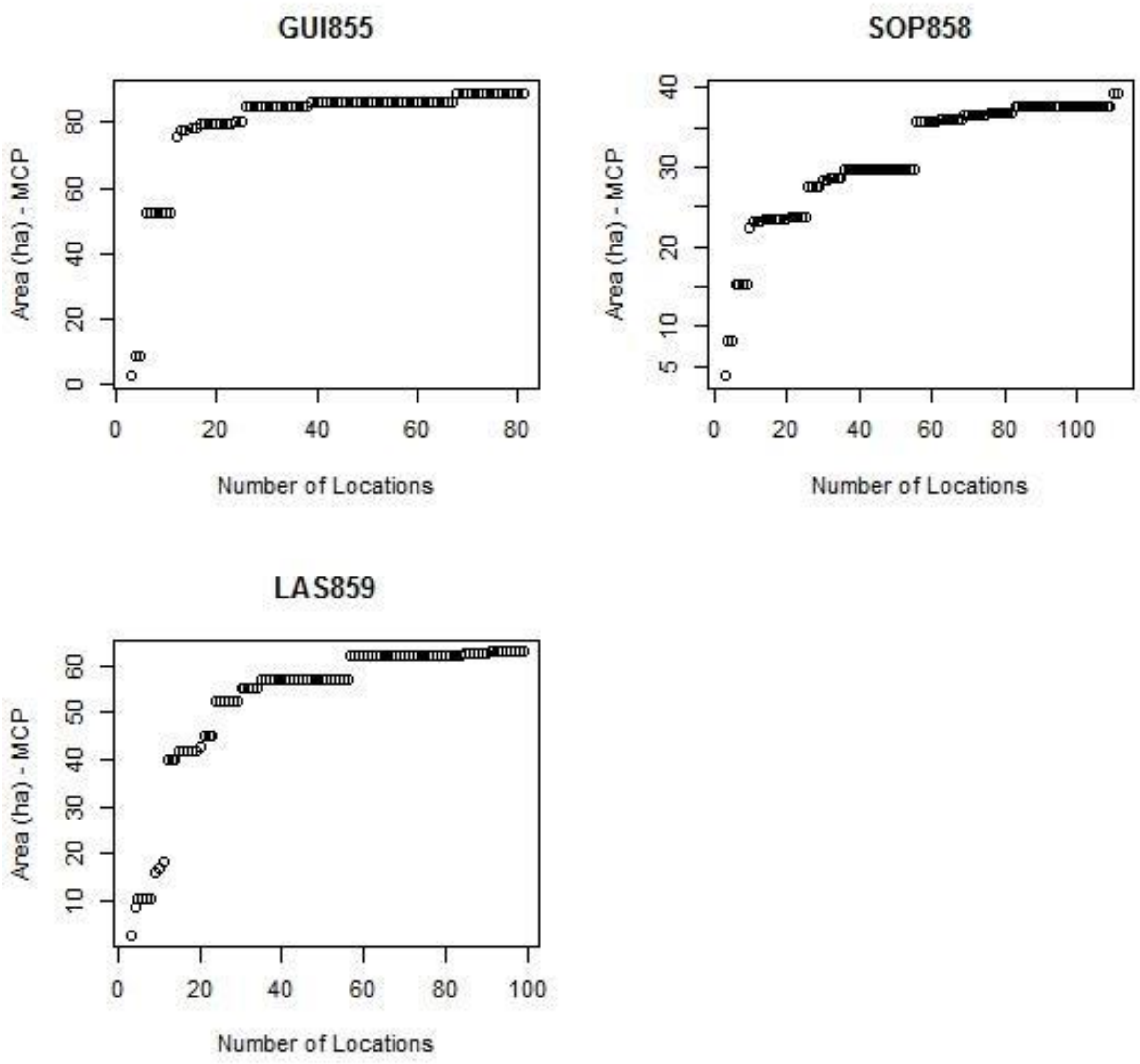

Fig. A.4. Site 1 relative frequencies of activities (foraging, roosting and resting) of BTF classified in three periods of the day (morning, midday and afternoon).

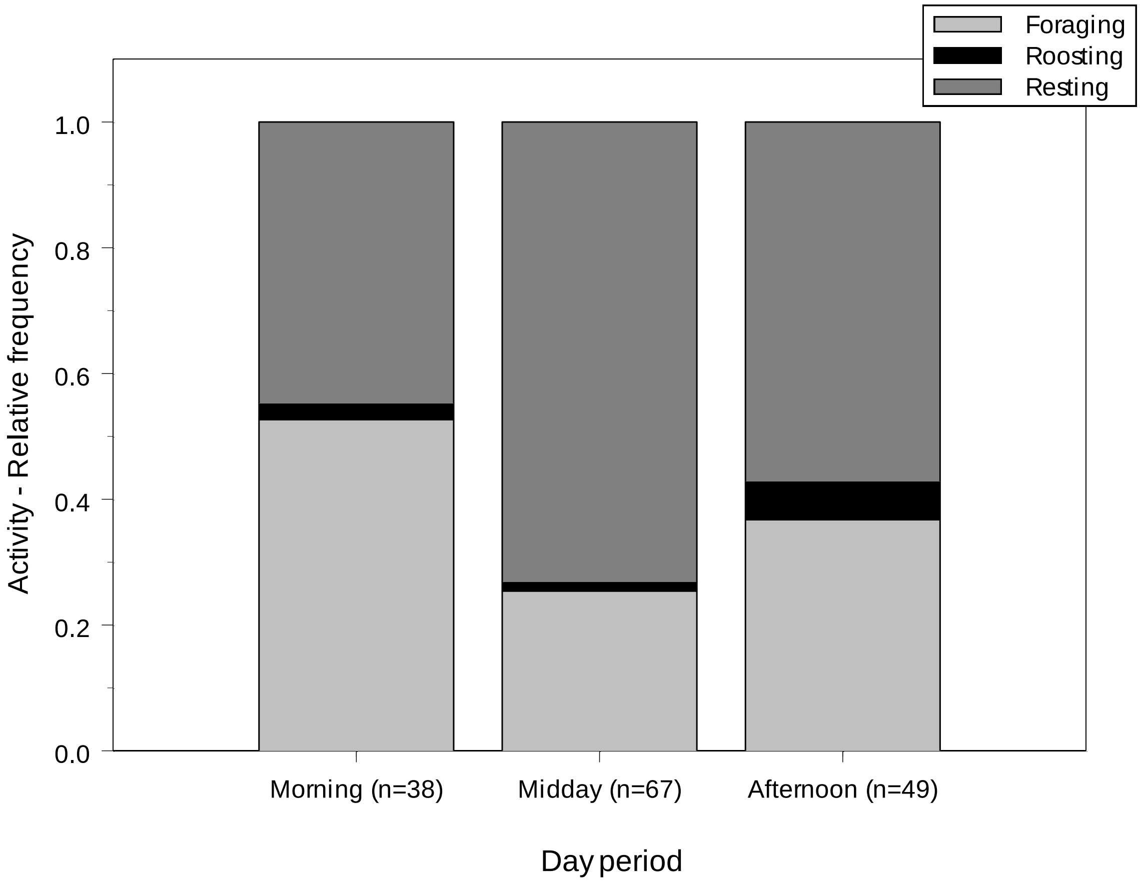

Fig. A.5. Site 2 relative frequencies of activities (foraging, roosting and resting) of BTF classified in three periods of the day (morning, midday and afternoon).

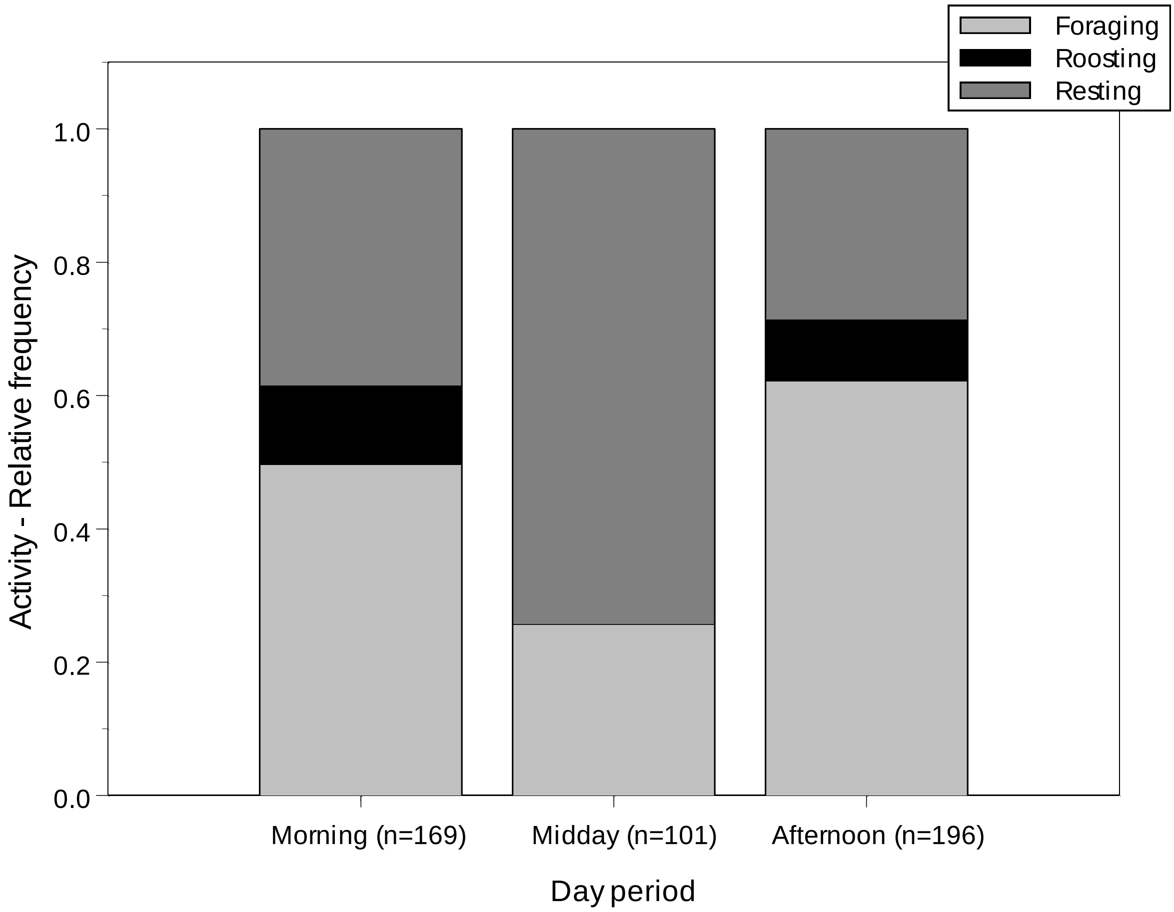

Fig. A.6. Box plot of BTF flock size in different periods of the day and different activities.

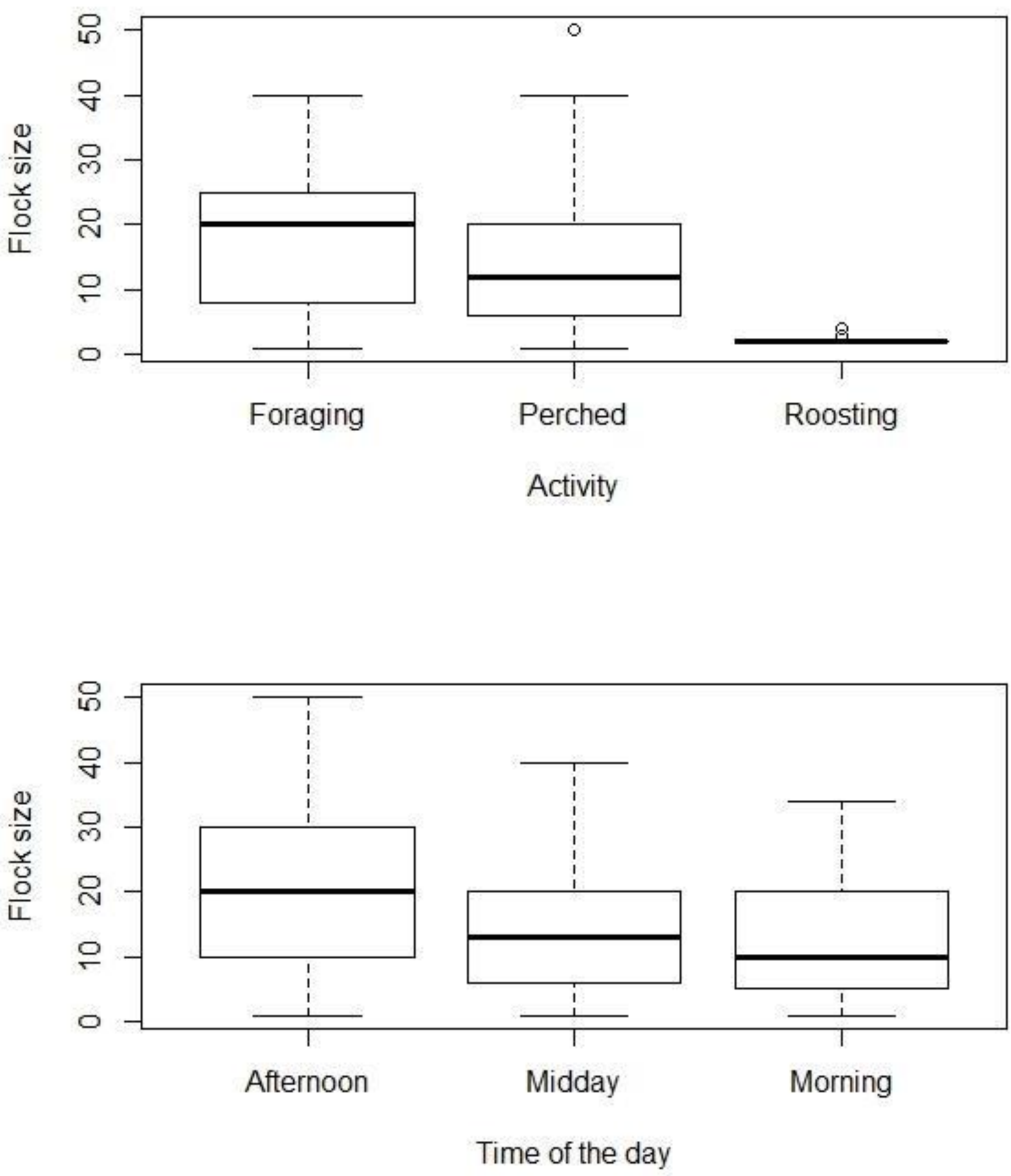

Fig. A.7. Flock size of BTF southern subspecies circle map for site 01.

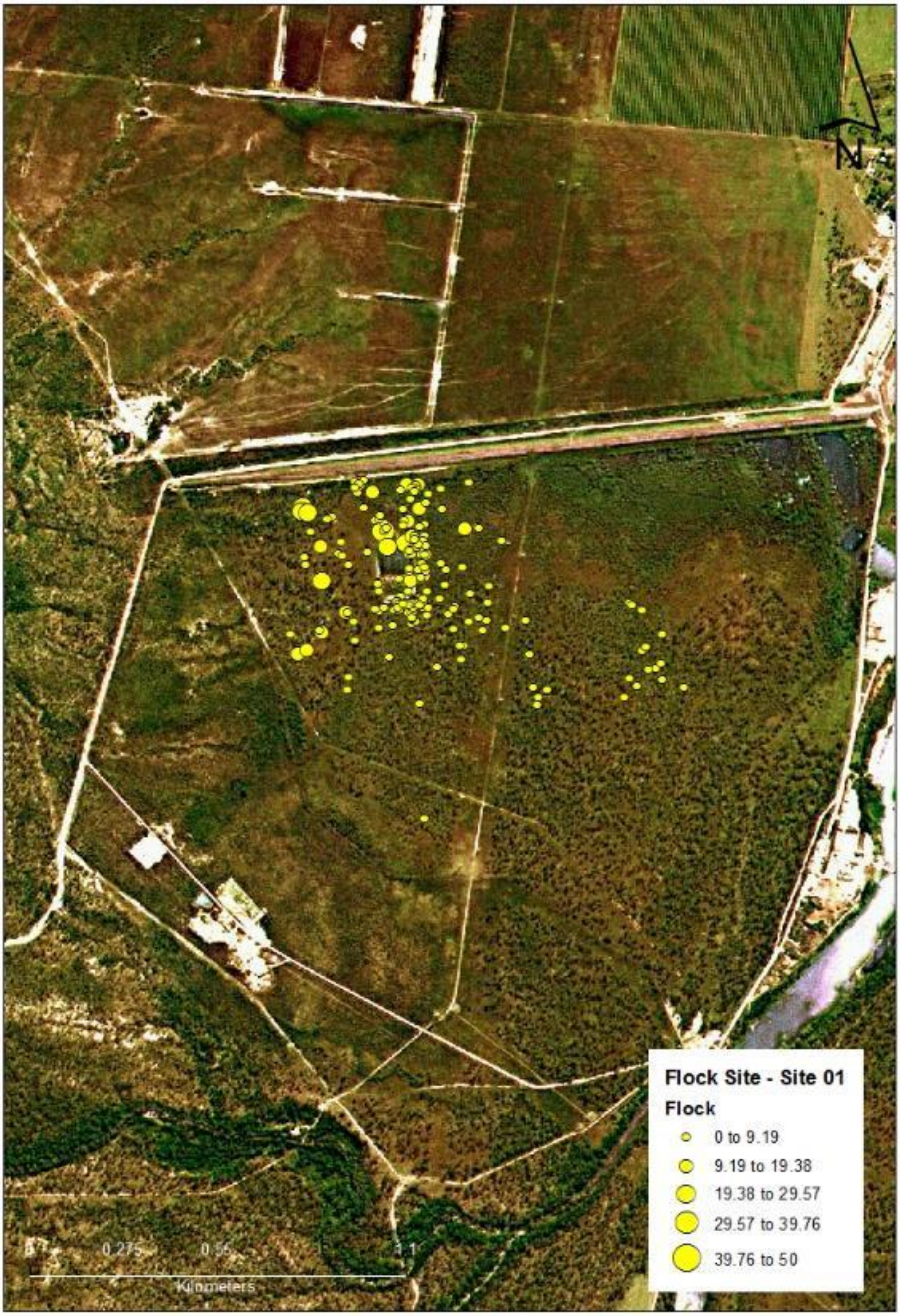

Fig. A.8. Flock size of BTF southern subspecies circle map for site 02.

Appendix B

Additional tables and figures from Chapter 3

Table B.1. Description of land use in the surveyed sites around Ross River dam.

	Site	RE	Property status	Activity
1	CW	Non-Remnant	Public	Historically lightly grazed (approx. 15 head) with minimal if any fires prior to 2008. Nowadays grazing is unofficial as cattle regularly get in from the neighboring property (often between 3 and 30 head). Fire reported in 2011. There was an attempt of early season patchy burn in 2012. *Hymenachne* spp. is the main weed ranger focus on annually. *Ziziphus mauritiana*, *Lanatana camara* and *Cryptostegia grandiflora* were treated with herbicide in 2014.
2	DBD	Non-Remnant	Private	Overgrazed, extensive patches eroded; presence of cattle is constant. BTF were never sighted. Sightings of BTF were reported close to the area.
3	CT	Non-Remnant	Private	Overgrazed, some patches eroded; presence of cattle is constant. BTF were sighted in the area drinking.
4	AC	11.3.12	Public	No official grazing from at least 1998 – maybe even before that. Unofficially there was heavy grazing of the foreshore within this site at times until 2009. Intense wildfire fire just prior to late 2012. A planned burn was conducted in 2011. Fire regime prior to 2011 is unknown. Annually, there are recruitment control *Hyparrhenia rufa*, Leucaena, *Lantana camara*, *Cryptostegia grandiflora*, *Z. mauritiana* and *Grewia asiatica* within this site. Comparatively, this site is characterized by the presence of native grasses and few weeds.
5	PF	11.3.35	Public	Moderately grazed annually. In sbout 2010 a hot fire went through this site. Weed control restricted to maintaining the firebreaks free of *Themeda quadrivalvis*. Some control of recruitment of *Z. mauritiana* by via basal braking with Access and diesel, and control of *T. quadrivalvis* throughout the paddock. Overall the site has big patches of *Z. mauritiana* and *Stylosanthes scabra*.
6	ATC	11.3.25b	Public	Heavy grazing until 2004 and an intense wildfire around the same year. Paddock destocked for a short period around 2004 and then restocked from 2004 -2009. There have been no stock in this paddock since 2009. Mechanical clearing of *Z. mauritiana* on large scale from 2010 to 2011, follow up by control of recruitment with diesel. In 2008 there was an attempt to foliar spray *T. quadrivalvis*. This site currently has the worst *T. quadrivalvis* infestation. Annually, a weed control spray is used in the paths/firebreaks to control *T. quadrivalvis*.
7	YD	11.3.35	Private	Before 2011: relatively low stocking rates but 'hot' fire. A lot of the native vegetation was cleared just prior 2011, significantly modifying BTF habitat. The former owner, around 2005/2006, kept stocking rates very high. There was a fire reported in 2011. Nowadays there are extensive patches with *Lantana camara* and the area is overgrazed. The site used to be a reliable area for BTF but records became scare over time. While conducting this study no BTF were sighted at this site. There were some clearing in the area as well.
8	SC	11.3.12	Public	It was grazed until late 2009, stocking rate was initially heavy, but wound back in 2004. Haven't been too many fires in that area (none in the last 7 years at least). *Z. mauritiana* control has being undertaken since 2009. Grader grass, *Hyparrhenia rufa* and *Ricinus communis* are regularly controlled.
9	MO	11.3.35	Private	The area is currently use for cattle and overall there is a layer of grasses all over the property with eroded areas only around the dam. No overgrazing.
10	MC	Non-Remnant	Private	Site used for recreational purposes. The area is mostly cleared but some large of *Eucalyptus plataphyla* and *Corymbia tessalaris* remain in the area. Due to the recreational activities, there are extensive patches of bare ground.

(Moroney, S., Warner, S. and Hunt, R. provided information about public sites and landholders on private sites).

Table B.2. List of grasses, forbs and sedges recorded during surveys in ten different sites in the vicinity of Ross River dam, south Townsville, Queensland. The ‘X’ represents the sites where the species was recorded.

	CW	DBD	CT	AC	PF	ATC	YD	SC	MO	MC	N Sites
Abildgaardia ovata				X				X			2
Alysicarpus vaginalis										X	1
Alternanthera		X	X		X	X		X			5
Aristida warburgii		X		X	X				X		4
Bothriochloa bladhii	X										1
Bothriochloa pertusa										X	1
Boerhavia spp	X		X								2
Bulbostylis barbata					X						1
Cajanus reticulatus				X							1
Crotalaria montana				X							1
Crotalaria medicaginea				X							1
Chrysopogon fallax	X			X							2
Cynodon dactylon	X		X		X					X	4
Cyperus iria					X						1
Chloris spp			X		X						2
Dactyloctenium aegyptium			X								1
Dichanthium			X			X				X	3
Digitaria			X	X							2
Epaltes australis										X	1
Eragrostis spp	X			X	X			X	X	X	6
Echinochloa	X		X								2
Eleocharis philippinensis	X										1
Fimbristylis				X			X				2
Glinus spp			X							X	2
Glycine tomentella							X			X	2
Grewia retusifolia							X				1
Heliotropium indicum								X			1
Hygrophilia angustifolia	X										1
Hyptis suaveolens		X		X	X		X	X	X	X	7
Heteropogon contortus	X	X	X	X	X	X	X	X	X	X	10
Heteropogon triticeus				X	X			X			3
Ischaeumum spp	X	X			X	X					4
Indigofera			X								1
Lomandra multiflora							X				1
Macroptilium lathyroides										X	1
Mecardonia procumbens	X				X						2
Marsilea	X										1
Melinis repens					X						1
Mnesithea rottboellioides				X							1
Neptunia gracilis									X		1
Panicum spp	X			X	X	X		X			5
Phyllanthus	X									X	2
Paspalum scrobiculatum	X			X							2
Passiflora					X			X			2
Portulaca pilosa										X	1
Stachytarpheta jamaicensis		X		X	X	X	X	X		X	7
Seedling	X	X									2
Stylosanthes scabra	X	X	X	X	X	X		X		X	8
Stylosanthes visosa		X						X	X		3
Sporobulus jacquemontii	X		X	X		X		X			5
Sida cordofolia	X	X	X		X		X		X		6
Tephrosia juncea				X			X				2
Themeda triandra				X			X		X		3
Themeda quadrivalvis						X					1
Urena lobata							X				1
Urochloa mosambiquensis			X		X						2
Urochloa subquadripara			X	X							2
Setaria surgens				X	X						2
Total species	19	10	16	22	20	9	11	13	8	15	

Table B.3. List of trees, shrubs and vines recorded during surveys in ten different sites in the vicinity of Ross River dam. The last column represents the total number of individuals of that species in all sites; the last line represents the number of species per site and the line before last represents arboreal abundance found per site.

	CW	DBD	CT	AC	PF	ATC	YD	SC	MO	MC	Total
Acacia spp	0	2	0	0	0	0	0	0	0	0	2
Acacia crassicarpa	0	0	0	0	0	0	66	0	0	0	66
Acacia holosericea	0	3	0	0	0	9	78	1	1	0	92
Acacia leptostachya	0	212	0	0	0	0	32	0	0	0	244
Acacia salicina	0	0	0	0	0	0	0	6	0	0	6
Bridelia leichhardtii	0	0	0	0	0	0	0	1	0	0	1
Bursaria spp	0	1	0	0	0	0	0	0	0	0	1
Canarium australianum	0	0	0	1	0	0	0	0	0	0	1
Capparis canescens	0	0	0	0	0	0	0	1	0	0	1
Cassia fistula	0	0	0	0	0	1	0	0	0	0	1
Casuarina cunning hamiana	0	0	0	0	0	2	0	0	0	0	2
Clerodendrum floribundum	0	2	0	0	0	0	0	0	0	0	2
Corymbia tessellaris	26	1	0	2	11	93	182	4	8	25	352
Corymbia clarksoniana	0	10	0	28	7	0	33	0	2	9	89
Corymbia erythrophloia	3	0	0	0	0	0	1	8	0	0	12
Corymbia dallachiana	28	1	34	75	26	0	8	46	0	0	218
Cryptostegia grandiflora	0	3	0	0	0	0	0	2	0	0	5
Dolichandrone heterophylla	0	4	0	0	8	0	0	0	0	0	12
Eucalyptus platyphylla	8	0	3	171	173	74	136	147	55	18	785
Eucalyptus drepanophylla	0	33	0	23	0	0	0	0	0	0	56
Ficus opposita	0	0	0	4	0	2	2	4	0	0	12
Flueggea virosa	0	0	0	0	0	0	0	12	0	0	12
Lantana camara	0	0	0	1	0	0	96	21	0	0	118
Larsenaikia ochreata	0	1	0	0	0	0	1	1	0	0	3
Lophostemon grandiflorus	2	66	0	6	0	35	0	21	0	0	130
Maytenus cunninghami	0	6	0	0	0	0	0	0	0	0	6
Melaleuca viridiflora	151	75	0	296	561	0	155	335	124	0	1697
Melaleuca nervosa	9	0	0	0	0	0	55	1	7	5	77
Melaleuca leucadendra	0	0	0	0	0	16	0	6	0	0	22
Millettia pinnata	0	0	0	0	0	3	0	4	0	0	7
Neolitsea brassii	0	0	0	0	0	8	0	0	0	0	8
Notelaea microcarpa	0	0	0	0	0	0	0	4	0	0	4
Pandanus spp	0	0	0	0	0	1	2	0	0	0	3
Planchonia careya	0	0	0	23	0	2	253	7	0	2	287
Pleiogynium timorense	0	0	0	0	0	4	0	0	0	0	4
Petalostigma pubescens	0	237	0	0	0	0	0	0	0	0	237
Psydrax alternata	0	2	0	0	0	0	0	0	0	0	2
Trophis scandens	0	0	0	0	0	0	0	1	0	0	1
Vachelia bidwillii	0	1	0	0	0	0	0	0	1	0	2
Grevillea striata	4	0	0	1	0	0	0	5	1	4	15
Xanthophyllum octandrum	0	0	0	0	0	8	0	0	0	0	8
Ziziphus mauritiana	8	6	0	0	43	150	1	154	0	3	365
Individual total	239	666	37	631	829	408	1101	792	199	66	4968
Species total	9	19	2	12	7	15	16	23	8	7	

Table B.4. Number of shrubs and trees (individuals and species) divided into four categories based on the diameter of breast height (DBH): shrubs, small trees (DBH <10 cm), medium trees 10-40 cm (DBH 10-40 cm) and large trees (DBH >40 cm). A total per transect and an overall total are also provided. Surveys were undertaken in 10 different sites in the vicinity of Ross River dam.

		Individual					Species				
		Shrubs	DBH<10	DBH10_40	DBH>40	Total	Shrubs	DBH<10	DBH10_40	DBH>40	Total
AC	Min	8	32	9	0	78	2	3	4	0	4
	Median	40	65	10	0	149	4	5	5	0	5
	Mean	39.4	76.6	9.8	0.4	126.2	3.6	4.4	5	0.4	5.4
	Max	74	134	11	2	168	5	5	7	2	7
	SD	30.88	38.72	0.84	0.89	41.9	1.1	0.9	1.22	0.9	1.5
ATC	Min	8	22	4	1	35	4	3	3	1	5
	Median	25	35	13	2	75	5	5	6	2	8
	Mean	28	40	11.4	2.2	81.6	4.8	5	5.8	2	7.6
	Max	50	62	18	4	118	6	7	9	4	10
	SD	15.44	18.12	5.2	1.3	36.2	0.8	1.6	2.77	1.2	2.1
CT	Min	0	0	1	0	1	0	0	1	0	1
	Median	0	0	2	0	3	0	0	2	0	1
	Mean	4	0.2	2.6	0.6	7.4	0.2	0.2	2.2	0.6	1.4
	Max	20	1	6	2	23	1	1	5	2	2
	SD	8.94	0.45	2.07	0.89	9.13	0.4	0.4	1.64	0.9	0.6
CW	Min	0	1	0	0	1	0	1	0	0	1
	Median	11	15	1	0	34	3	4	1	0	5
	Mean	8.6	35.6	3.4	0	47.8	2.2	4	2	0	4.8
	Max	17	129	13	0	158	4	7	6	0	7
	SD	8.14	53.14	5.4	0	63.8	2.1	2.1	2.34	0	2.3
DBD	Min	7	30	1	0	56	2	2	1	0	5
	Median	36	41	8	0	146	4	5	2	0	7
	Mean	70.4	53.8	9	0	133.2	3.8	4.8	2.6	0	7
	Max	188	105	21	0	246	5	6	5	0	9
	SD	76.74	30.01	7.97	0	77.3	1.1	1.6	1.52	0	1.4
MC	Min	0	0	2	0	6	0	0	2	0	3
	Median	3	1	7	1	12	2	1	4	1	4
	Mean	3	1.8	7.4	1	13.2	1.8	1	4.6	0.8	4
	Max	7	5	14	2	24	3	2	8	2	6
	SD	2.7	1.92	4.5	1	6.6	1.3	0.7	2.14	0.8	1.2
MO	Min	1	2	4	0	10	1	1	2	0	2
	Median	2	5	14	0	24	2	3	3	0	4
	Mean	10.2	16.6	12.8	0.2	39.8	2	2.6	3.4	0.2	3.8
	Max	33	62	20	1	109	4	4	5	1	6
	SD	13.9	25.5	6.4	0.4	40.9	1.2	1.1	1.1	0.4	1.5
PF	Min	23	51	2	0	99	3	3	2	0	4
	Median	44	105	3	0	131	3	5	3	0	5
	Mean	53.8	107.4	4.2	0.4	165.8	3.6	4.6	2.6	0.4	5
	Max	93	156	8	1	251	5	6	3	1	6
	SD	29.6	40.9	2.4	0.55	66.1	0.89	1.1	0.55	0.55	0.7
SC	Min	41	23	4	0	78	3	3	3	0	4
	Median	94	25	8	0	148	8	5	4	0	10
	Mean	89.6	60.8	7.6	0.4	158	7.6	5.4	4.2	0.4	9.4
	Max	119	142	11	1	269	12	9	6	1	14
	SD	32.5	53.9	2.51	0.5	75.8	3.6	2.3	1.1	0.5	4.3
YD	Min	54	23	13	1	100	3	1	6	1	4
	Median	65	116	39	4	236	6	5	8	3	10
	Mean	90	84.4	42	3.8	220.2	7	5.2	8.4	2.4	8.6
	Max	155	136	80	7	372	11	8	13	3	11
	SD	45.6	54.3	24.9	2.2	112.8	3.1	2.9	2.9	0.9	3.1
Total	Min	0	0	0	0	1	0	0	0	0	1
	Median	23	29	8	0	81	3.5	4	4	0	5
	Mean	39.7	47.72	11.02	0.9	99.36	3.66	3.72	4.08	0.72	5.7
	Max	188	156	80	7	372	12	9	13	4	14

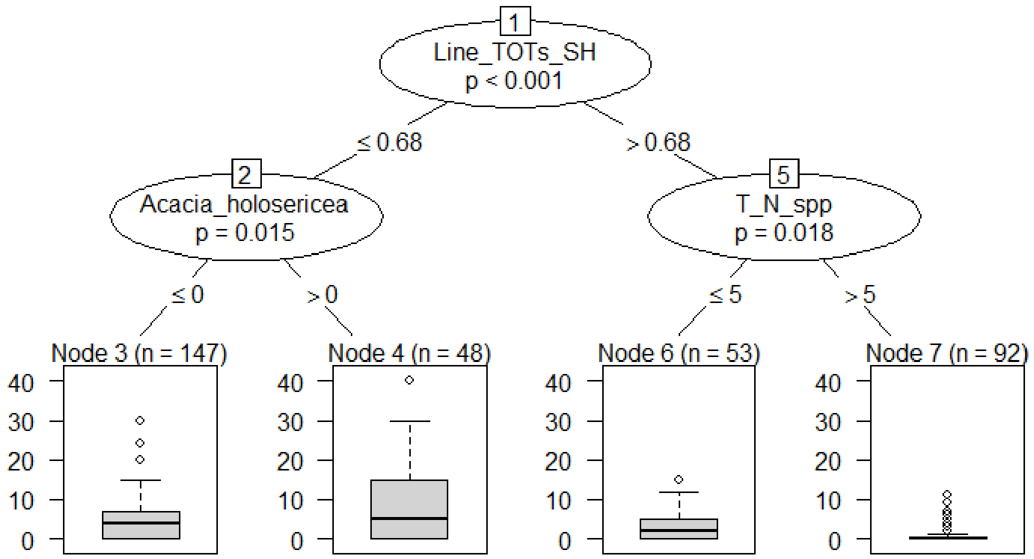

Fig. B.1. Conditional inference tree examining BTF flock size in different sites using woody plant cover structure and composition as explanatory variables (number of variables = 60). Flock size was best classified into four groups (leaves) considering this subset of data. Node 4 is primarily large flock, nodes 3 and 6 medium flocks and node 7 small flocks. Node 1: foliage projective cover of shrubs. Node 2: abundance of *Acacia holosericea*. Node 5: total arboreal richness.

Appendix C

Additional tables and figures from Chapter 4

Table C.1. Species of grasses or forbs (including the shrub *Stylosanthes scabra*) and the number of used habitats (nesting sites) of BTF were present. Grass/forbs species percentage compared in used and available areas. Nests = number of nesting habitats it was recorded. Sites = number of sites it was recorded from the available areas. (Variable codes: $\overline{x}$ = mean; SD= standard deviation; SE= standard error; Min= minimum value observed; Max= maximum value observed).

			Used					Available							
Species	Nests	Sites	$\overline{x}$	SD	SE	Min	Max	$\overline{x}$	SD	SE	Min	Max	t value	df	p value
Stylosanthes scabra	11	8	5.5	6.7	1.5	0	20	2.5	4.4	0.6	0	23	-1.7952	25.811	0.084
Heteropogon contortus	7	10	4.3	7.7	1.7	0	26.7	2.6	4.8	0.7	0	18.0	-0.9339	25.116	0.3593
Themeda triandra	7	3	4.8	8.3	1.8	0	25	1.6	4.5	0.6	0	25	-1.6197	23.717	0.1185
Eragrostis spp	5	6	1.5	3.9	0.9	0	16.7	0.7	1.7	0.2	0	6.4	-0.8949	21.87	0.3806
Passiflora spp	4	2	1.2	2.8	0.6	0	10	0.03	0.1	0.02	0	1	-1.9354	19.046	0.0679
Aristida warburgii	3	4	2.6	6.5	1.4	0	21	2.04	6.1	0.8	0	29	-0.3513	32.945	0.7276
Sporobulus jacquemontii	3	5	0.8	2.3	0.5	0	8.3	0.2	0.9	0.1	0	5.0	-1.0731	21.164	0.2953
Stachytarpheta jamaicensis	3	7	1.2	3.8	0.8	0	13.3	3.5	7.3	1.03	0	32.2	1.634	62.9	0.1073
Cynodon dactylon	2	4	3.1	11.9	2.7	0	53.3	0.7	2.6	0.4	0	14.6	-0.87	19.723	0.3948
Alysicarpus vaginalis	1	1	0.7	3.3	0.7	0	15	0.1	0.8	0.1	0	6	-0.8295	19.98	0.4166
Bothriochloa spp.	1	1	0.25	1.1	0.25	0	5	0.18	0.9	0.13	0	6	-0.2472	30.346	0.8064
Chrysopogon fallax	1	2	0.3	1.5	0.3	0	6.7	1.1	3.3	0.5	0	18	1.2707	67.047	0.2082
Heteropogon triticeus	1	3	0.2	0.7	0.2	0	3.3	0.2	0.7	0.1	0	3.0	0.4196	33.505	0.6775
Ischaeumum spp	1	4	0.08	0.4	0.08	0	1.7	1.1	4.1	0.6	0	24.4	1.7256	51.006	0.0905
Phyllanthus spp	1	2	0.4	1.9	0.4	0	8.3	0.08	0.4	0.06	0	3.0	-0.7989	19.871	0.4338
Portulaca pilosa	1	1	0.08	0.4	0.08	0	1.7	0.08	0.6	0.08	0	4.0	-0.0303	52.679	0.976

Table C.2. Comparison of tree density in used and available sites. (Variable codes: x̄ = mean; SD= standard deviation; SE= standard error; Min= minimum value observed; Max= maximum value observed).

Species	**Used**					**Available**					**t value**	**df**	**p-value**
	x̄	SD	SE	Min	Max	x̄	SD	SE	Min	Max			
Acacia crassicarpa	0	0	0	0	0	13.2	83.8	11.8	0	590	1.1135	49	0.2709
Corymbia tessellaris	22.4	45.4	10.1	0	131.5	70.4	210.3	29.7	0	1450.0	1.5284	59.008	0.1318
Corymbia clarksoniana	2.6	11.8	2.6	0	52.6	17.8	46.6	6.6	0	270.0	2.139	61.825	0.03639
Corymbia dallachiana	223.6	680.9	152.3	0	2788.5	43.6	78.4	11.1	0	270.0	-1.1791	19.202	0.2528
Eucalyptus platyphylla	77.6	79.8	17.8	0	289.4	157.0	185.7	26.3	0	810.0	2.5005	67.55	0.01484
Eucalyptus drepanophylla	0	0	0	0	0	11.2	36.1	5.1	0	190.0	2.1926	49	0.03312
Lophostemon grandiflorus	3.9	17.6	3.9	0	78.9	26.0	81.3	11.5	0	410.0	1.814	59.105	0.07476
Melaleuca viridiflora	185.5	313.5	70.1	0	1104.9	339.4	571.2	80.8	0	2300.0	1.4392	61.148	0.1552
Melaleuca nervosa	0	0	0	0	0	15.4	64.3	9.1	0	440.0	1.6925	49	0.0969
Melaleuca leucadendra	2.6	11.7	2.6	0	52.1	4.4	17.1	2.4	0	100.0	0.4959	50.49	0.6221
Planchonia careya	3.9	12.9	2.9	0	52.6	57.4	169.7	24.0	0	790.0	2.211	50.393	0.0316
Petalostigma pubescens	1.31	5.9	1.31	0	26.3	47.4	186.7	26.4	0	820.0	1.7434	49.243	0.08751
Grevillea striata	6.6	14.5	3.2	0	52.6	3.0	7.6	1.07	0	40.0	-1.0486	23.344	0.3051
Ziziphus mauritiana	52.6	121.9	27.3	0	420.9	73.0	177.5	25.1	0	940.0	0.5502	50.731	0.5846

Appendix D

Additional tables and figures from Chapter 5

Table D.1. Grass, sedges and forb species recorded in used (US), neighbouring (NG) and available (AV) areas in BTF habitat. Codes: P = present; A= absent.

UsedAvail	US	NG	AV	Life form	An_Pern	Status
Abildgaardia ovata	A	0	6	Grass	Perennial	Native
Alternanthera spp	A	P	25	Forb	Annual	Unknown
Alysicarpus vaginalis	P	P	6	Forb	An_Pe	Exotic
Aristida warburgii	A	P	P	Grass	Perennial	Native
Bacopa floribunda	P	A	A	Forb	An_Pe	Native
Boerhavia spp	A	A	P	Forb	An_Pe	Native
Bothriochloa spp	P	P	P	Grass	Perennial	Exotic
Bulbostylis barbata	A	A	P	Sedge	Annual	Native
Cajanus reticulatus	A	A	P	Forb	Perennial	Native
Centipeda nidiformis	P	P	A	Forb	Perennial	Native
Chloris spp	A	A	P	Grass	An_Pe	Exotic
Chrysopogon fallax	A	P	P	Grass	Perennial	Native
Crotalaria spp	A	A	P	Forb	An_Pe	Native
Cynodon dactylon	P	P	P	Grass	Perennial	Exotic
Cyperus iria	A	A	P	Sedge	Perennial	Native
Dactyloctenium aegyptium	A	A	P	Grass	An_Pe	Exotic
Dichanthium spp	A	P	P	Grass	Perennial	Unknown
Digitaria ciliaris	A	P	P	Grass	An_Pe	Unknown
Echinochloa spp	A	A	P	Grass	An_Pe	Unknown
Eleocharis philippinensis	P	P	P	Grass	Perennial	Native
Epaltes australis	A	A	P	Forb	Annual	Native
Eragrostis spp	P	P	P	Grass	An_Pe	Unknown
Fimbristylis spp	A	A	P	Sedge	An_Pe	Native
Galactia tenuiflora	P	A	A	Forb	Perennial	Native
Glinus spp	A	A	P	Forb	Annual	Native
Glycine tomentella	A	A	P	Forb	Perennial	Native
Grewia retusifolia	A	A	P	Shrub	Perennial	Native
Heliotropium indicum	A	A	P	Forb	Annual	Exotic
Heteropogon contortus	P	P	P	Grass	Perennial	Native
Heteropogon triticeus	A	A	P	Grass	Perennial	Native
Hygrophilia angustifolia	A	A	P	Forb	Annual	Native
Hyptis suaveolens	P	P	P	Forb	Annual	Exotic
Indigofera spp	A	A	P	Shrub/Forb	An_Pe	Unknown
Ischaeumum spp	P	P	P	Grass	Annual	Native
Lomandra multiflora	A	A	P	Forb	Perennial	Native
Ludwigia hyssopifolia	P	P	A	Forb	Annual	Exotic
Macroptilium lathyroides	P	P	P	Forb	An_Pe	Exotic
Marsilea spp	A	A	P	Forb	Perennial	Native
Mecardonia procumbens	P	P	P	Forb	Perennial	Exotic
Melinis repens	A	A	P	Grass	An_Pe	Exotic
Mnesithea rottboellioides	A	A	P	Grass	Perennial	Native
Neptunia gracilis	A	A	1	Forb/Shrub	Perennial	Native
Panicum spp	P	P	P	Grass	Perennial	Native
Paspalum scrobiculatum	P	P	P	Grass	Annual	Native
Passiflora foetida	A	P	P	Vine	An_Pe	Unknown
Phyllanthus spp	P	P	P	Forb/Shrub	An_Pe	Unknown
Portulaca pilosa	6	P	P	Forb	Annual	Exotic
Rostellularia adscendens	P	A	A	Forb	An_Pe	Native
Setaria spp	P	P	P	Grass	Annual	Native
Sida cordofolia	P	P	P	Forb	Perennial	Exotic
Sporobulus spp	P	P	P	Grass	Perennial	Exotic
Stachytarpheta jamaicensis	P	P	P	Forb	Perennial	Exotic
Stylosanthes scabra	P	P	P	Shrub	Perennial	Exotic
Stylosanthes visosa	A	A	P	Forb	Perennial	Exotic
Tephrosia juncea	A	A	P	Forb	An_Pe	Native
Themeda quadrivalvis	A	A	P	Grass	Annual	Exotic
Themeda triandra	P	P	P	Grass	Perennial	Native
Urena lobata	A	A	P	Forb	Annual	Exotic
Urochloa mosambiquensis	A	P	P	Grass	Perennial	Exotic
Urochloa subquadripara	A	A	P	Grass	Annual	Exotic
Zornia areolata	P	A	A	Forb	Perennial	Native

Table D.2. Trees and shrubs with abundance of more than 15 individuals in used and available areas of BTF.

Parameters	Used					Available					t value	df	p-value
	$\bar{x}$	SD	SE	Min	Max	$\bar{x}$	SD	SE	Min	Max			
Acacia crassicarpa	0	0	0	0	0	13.2	83.8	11.8	0	590	1.1135	49	0.271
Acacia holosericea	36.7	106.4	18.5	0	447.2	18.4	68.7	9.7	0	350	-0.8733	49.569	0.387
Acacia leptostachya	0	0	0	0	0	48.8	230.4	32.6	0	1490	1.4979	49	0.141
Corymbia tessellaris	19.1	66.8	11.6	0	368.3	70.4	210.3	29.7	0	1450	1.6051	62.884	0.114
Corymbia clarksoniana	5.6	17.1	2.9	0	78.9	18.8	46.6	6.6	0	270	1.6909	66.792	0.095
Corymbia dallachiana	20.7	50.4	8.8	0	210.4	43.6	78.4	11.1	0	270	1.618	80.968	0.1096
Eucalyptus platyphylla	61.4	124.7	21.7	0	605.1	157	185.7	26.3	0	810	2.8062	80.959	0.0063
Eucalyptus drepanophylla	5.6	19.5	3.4	0	78.9	11.2	36.1	5.1	0	190	0.9168	78.37	0.362
Lantana camara	0	0	0	0	0	23.6	83.2	11.8	0	420	2.0064	49	0.0503
Lophostemon grandiflorus	0.8	4.6	0.8	0	26.3	26	81.3	11.5	0	410	2.1863	49.47	0.0335
Melaleuca spp.	378.7	544.2	94.7	0	1946.7	354.8	584.9	82.7	0	2300	-0.1897	72.042	0.8501
Petalostigma pubescens	0.8	4.6	0.7	0	26.3	47.4	186.7	26.4	0	820	1.7643	49.089	0.0839
Planchonia careya	0	0	0	0	0	57.4	169.7	24.0	0	790	2.3912	49	0.0207
Ziziphus mautitiana	3.9	16.3	2.8	0	78.9	73	177.5	25.1	0	940	2.7325	50.244	0.0086

Appendix E

Addressing potential cumulative impacts of development on threatened species: the case of the endangered Black-throated finch

Introduction

The high rates of biodiversity decline documented globally demand close attention to the conservation status of threatened species where threats are ongoing (Butchart et al. 2010, Waldron et al. 2013). Many species continue to decline because of changes in land use that include broadscale land clearing for agriculture and urban development, as well as more subtle effects from fragmentation, invasive species, grazing, changed fire regimes and shifting climate envelopes (Brook et al. 2008).

Globally, offsets are seen as a mechanism for achieving no overall net loss or net environmental gains in the face of development pressures (see summaries and examples in ten Kate et al. 2004, Gibbons and Lindenmayer 2007, McKenney and Kiesecker 2010, Gardner et al. 2013, Maron et al. 2015). In Australia, where threatened species and ecosystems (e.g. matters of national environmental significance (MNES); EPBC 1999, Australian Government 2012) are likely to be adversely affected by development, state and federal legislation (e.g. Australian Government 2012, EHP 2013, Queensland Government 2014b) dictates that development proponents must show that impacts will be offset. This may mean compensating for MNES losses to achieve no net loss (Australian Government 2012, Queensland Government 2014i) by implementing management to maintain or improve viability and provide equivalence for the MNES (Queensland Government 2014j, b). However, it is difficult to compensate for loss of threatened species or their habitat; particularly for species with very specific requirements, species' whose requirements are poorly known, or where suitable habitat cannot readily be recreated (Pilgrim et al. 2013). Thus, avoiding loss of the habitat of threatened species to achieve the goals of the acts, legislation and policies listed above is logistically problematic (Gibbons and Lindenmayer 2007), and evaluating whether this loss has been avoided is equally fraught (Morris et al. 2006, Bedward et al. 2009, Maron et al. IN PRESS). Creating offsets that reliably avoid loss of threatened species habitat, particularly when known areas of high habitat suitability are compromised, lacks both theoretical support and practical methodology for the majority of species.

It is well recognised that potential cumulative impacts (Franks et al. 2010) are often overlooked when multiple developments are considered in isolation and the landscape context is ignored (Quetier and Lavorel 2011), decreasing the likelihood of achieving a "no net loss" outcome. Defining appropriate reference frames and counterfactual scenarios is critical to effective offsetting (Bull et al. 2014, Virah-Sawmy et al. 2014) as are timeframes for establishing (Gibbons and Lindenmayer 2007) and maintaining the benefits of the offsets (Virah-Sawmy et al. 2014). We argue that there is insufficient land that is BTF habitat available for offsetting, even though this approach to offsetting has been criticised as being logically flawed (Bekessy et al. 2010). We also argue that there is insufficient land available for an offsetting scheme where restoration or habitat improvement is the mechanism used to create the offsets if all the planned developments go ahead. A precautionary approach is critical because "when offsetting is proposed, impacts to biodiversity are certain and effective offsets are not" (Norris 2014). Both statutory (strategic assessments; Australian Government 2014b) and non-statutory (e.g. Galilee Basin Offset Strategy; EHP 2013) instruments exist to help plan within a landscape context. Four strategic assessments are in place and twelve in progress across Australia (Australian Government 2013, 2014b). However, there is no requirement to coordinate offset selection, tenure arrangements and maintenance of offsets outside of strategic assessment frameworks.

We examine the extent of potential cumulative impacts of mining industry activities in the case study of a threatened bird, the black-throated finch *Poephila cincta cincta* (hereafter BTF). To date, there has been no investigation into whether there would be sufficient habitat left and available to be used as offset, should all the planned mining activities proceed. Without this investigation, there can be no real understanding of the probability of persistence of BTF. This is symptomatic of development procedure in Australia and elsewhere, so we aim to use the case of the BTF to highlight the importance of investigating cumulative impacts on threatened species more generally.

BTF inhabit open woodlands and forests with seeding grasses and free water (summarised in Higgins et al. 2006b) and generally take their food, primarily grass seeds, from the ground (Zann 1976b). Most aspects of BTF social structure and breeding behaviour in the wild are poorly known (Higgins et al. 2006b). Nests are usually in trees, shrubs, mistletoes, raptor nests or tree hollows (Zann 1976b, Higgins et al. 2006b) and clutch size varies from three to nine (Higgins et al. 2006b). Patterns of movement across the landscape, if they occur, are poorly known (Queensland Government 2006b) although two sightings of one banded bird were 16 km apart (Rechetelo et al. submitted). There is no evidence for larger scale movements across the BTF's range.

The extent of occurrence of BTF has contracted by an estimated 80% over the last 30 years (Higgins et al. 2006b, BTFRT 2007b). Declines have been linked to land use change, primarily land clearing and intensification of livestock grazing (Garnett and Crowley 2000a, Higgins et al. 2006b, BTFRT 2007b, Garnett et al. 2011b), conversion of habitat to pasture, including the introduction of non-native fodder grasses, fragmentation, weed invasion, urban expansion and synergistic effects involving these factors, drought and fire (Garnett and Crowley 2000a, BTFRT 2007b, Garnett et al. 2011b). These forces have occurred in an area that has undergone "one of the most rapid landscape transformations ever documented" (see Seabrook et al. 2006 and references therein). Within BTF range in Queensland, vegetation clearing has been relatively more extensive in the south than the north (DSITIA 2014) and past clearing has been coincident with BTF declines (compare Franklin 1999, Seabrook et al. 2006, BTFRT 2007b, Department of Environment and Resource Management 2009), especially in the Brigalow Belt bioregion where over 60% of the original vegetation has been cleared (Seabrook et al. 2006, Seabrook et al. 2007). BTF are not known to have declined within the northeast Desert Uplands, an area subject to relatively less land clearing than other parts of BTF range (DSITIA 2014). There are extensive resource extraction leases and exploration permits in this region (Queensland Government 2015), representing a hitherto unrealised threat to BTF. Planned and approved developments include large open-cut mines (e.g. 24 km long (Hancock Prospecting Pty Ltd 2010a)), underground mines subject to post-mine subsidence (Macmines Australia Pty Ltd 2012, SGCP 2012b) and associated infrastructure.

Because of its endangered status, offsets are being considered in the face of resource extraction developments planned in remaining BTF habitat (e.g. Eco Logical Australia 2012a, EHP 2013). However, each development is considered individually, and there has been no attempt to estimate the potential impacts on BTF of all the developments in concert. We examine the total footprint of all the resource extraction and exploration leases in BTF habitat and from this we identify the amount of BTF habitat available that could be used for offsetting purposes.

Methods

Species data

The black-throated finch is an Australian Estrildid grass-finch consisting of two subspecies, the northern: *Poephila cincta atropygialis*, confined to Cape York Peninsula and the northern and western Gulf Plains, QLD; and the southern subspecies: *Poephila cincta cincta*, now largely restricted to the Townsville Plain and portions of the Desert Uplands and Brigalow Belt bioregions, QLD, south to about 23^{o}S (Figure 1). The northern subspecies is not listed as threatened, while the southern subspecies is listed as endangered under the Federal *Environment Protection and Biodiversity Conservation Act* (EPBC 1999), the *QLD Nature Conservation Act* (Queensland Government 2006b) and the New South Wales *Threatened Species Conservation Act* (New South Wales Government 1995a). BTF once occurred as far south as 31^{o}S in New South Wales (NSW), but there are no recent records and it may now be extinct in NSW (Higgins et al. 2006b). In the vast area of its former range in southern QLD (south of 23^{o}S) there have been only nine records since 1980 (Barrett et al. 2003; QLD Wildnet and Black-throated Finch Recovery Team databases, unpublished data), including one in 1990 from near Stanthorpe in the extreme south and one at Rockhampton in 2004 (Figure 1). BTF now appear to have two major strongholds – parts of the Townsville Plain in the northern Brigalow Belt bioregion, and along the eastern edge of the Desert Uplands bioregion (see Environment Australia 2000).

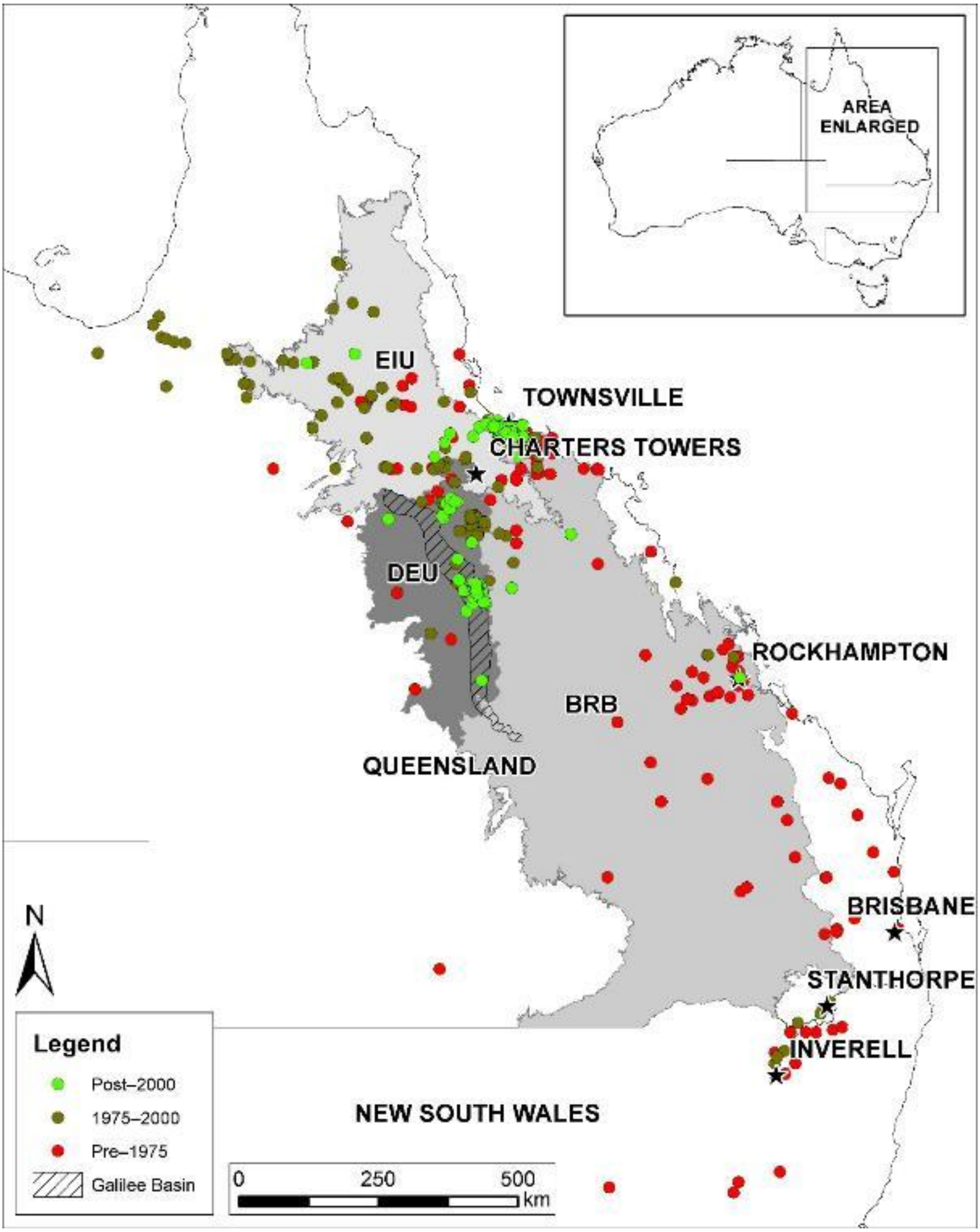

Fig. 1. Distribution of BTF records colour-coded by years. BTF records from all year-classes were used to create the bioclimatic model. The most relevant IBRA bioregions are shaded: BRB = Brigalow Belt; DEU = Desert Uplands; EIU = Einasleigh Uplands. Relevant towns and the approximate extent of the Galilee Basin coal measure (from Queensland Government 2011) are shown.

BTF records were accessed from Birdlife Australia (Blakers et al. 1984, Barrett et al. 2003), CSIRO Ecosystem Sciences (Perry et al. 2012), the Black-throated Finch Recovery Team (BTFRT, unpublished data, 2014) and ongoing research (Rechetelo et al. submitted), environmental impact statement data available on the internet (GHD 2012, 2013, 2014b), and data acquired through a 10-year water source monitoring program co-ordinated by the BTFRT on the Townsville Plain. Data were vetted by eliminating northern subspecies records, plus any listed as unspecified subspecies north of 18.9^{o} S, which is the approximate northern limit of the southern subspecies. Three BTF records that fell in the Pacific Ocean were assumed to be errors and removed. Five records of unspecified subspecies BTF from 1880-1928 in the western edge of the species' range (north west Queensland) were removed because they were surrounded by more contemporary northern subspecies records and literature suggests only the northern subspecies occurs that far west (Zann 1976b). In order to account for spatial bias in the occurrence data (Kramer-Schadt et al. 2013) because of unequal sampling effort (largely a result of proximity to an urban centre), we employed the following two spatial thinning methods: the vetted BTF records were rounded to both two and three decimal places, and any duplicate locations removed after these rounding processes. These roundings amount to roughly 700 m and 70 m, respectively, across BTF range. Rounding left us with totals of 466 and 850 BTF record locations for two and three decimal places, respectively. While rounded duplicates were removed, the original precision was retained in the remaining occurrence data for running the models. The BTF records shown in Figure 1 are the three decimal place rounded records. The two decimal place records appear the same at the scale shown in Figure 1.

Climate data

Climate data were derived from ANUCLIM (McMahon et al. 1995) at a 9-second resolution (approximately 250 m grids). The climate variables used were 30-year averages for the period 1976-2005 of annual mean temperature, temperature seasonality, maximum temperature of the warmest period, minimum temperature of the coldest period, annual precipitation, precipitation of the wettest period, precipitation of the driest period and precipitation seasonality. These variables have been shown to represent the bioclimatic limits of vertebrates throughout this region (VanDerWal et al.

2009a, VanDerWal et al. 2009b, Williams et al. 2010, Reside 2011, Anderson et al. 2012, Reside et al. 2012a, Reside et al. 2012b, Reside et al. 2013, VanDerWal et al. 2013).

Modelling protocol

The BTF climate envelope was modelled using Maxent (Phillips et al. 2006). In addition to rounding occurrence data described above, to further account for spatial bias in the occurrence records, we used target-group background points in Maxent (Phillips and Dudik 2008). Background points were derived from records of common mynas (*Acridotheres tristis*) and six sympatric woodland species: weebills (*Smicrornis brevirostris*), squatter pigeons (*Geophaps scripta*), peaceful doves (*Geopelia striata*), double-barred finches (*Taeniopygia bichenovii*), black-faced woodswallows (*Artamus cinereus*) and little woodswallows (*A. minor*), obtained from the online Atlas of Living Australia (Atlas of Living Australia), systematic data collection sites (Perry et al. 2012), standardised water source counts and camera trap observation points (GHD 2013, 2014a). The species in the the target-group background were chosen because they: are strongly associated with humans (common myna (Higgins et al. 2006b)), and thus likely to account for spatial bias of species records around urban centres and along roads (Yackulic et al. 2013); are widespread and common over much of Australia (weebill, peaceful dove (Higgins and Davies 1996, Pizzey and Knight 1997)); share a similar but not so dramatic pattern of decline to BTF (squatter pigeon (Higgins and Davies 1996)); are granivores, often occurring in similar areas and habitat to BTF (squatter pigeon, peaceful dove, double-barred finch (Higgins and Davies 1996, Pizzey and Knight 1997, Higgins et al. 2006b)); are known to associate with BTF in mixed flocks (weebill, woodswallows, double-barred finch (Higgins et al. 2006b, Vanderduys et al. 2012)); are of a similar size to or smaller than BTF (double-barred finch, weebill (Pizzey and Knight 1997, Higgins et al. 2006b)) and often cryptic (weebill). By using the species above, and clipping target-group background points to QLD and NSW, we addressed the tension between overly generous environmental space represented in the background points and environmental space too similar to that of BTF occurrences (VanDerWal et al. 2009a). Background points were rounded to three decimal places and any duplicate locations removed after this rounding process. However, after rounding and thinning, the original precision was retained. This left us with

99064 background points. Maxent used 99100 and 99334 points (background plus BTF presence points) to create the two decimal and three decimal place distribution models, respectively.

We ran the model twice: first with the two decimal place occurrence records, and second with the three decimal place occurrence records. This allowed us to investigate the trade off between levels of spatial thinning, versus sample size. Initially, the suitable climate space for BTF was designated as the area within the two and three decimal place Maxent-derived climate envelopes that had suitability scores ≥0.5. Both models were clipped, so that below the threshold suitability became zero, and above the threshold was assumed to be climatically suitable. We validated the models by area under the receiver operating characteristic curve (AUC) (Swets 1988).

Because much of BTF range is within vast tropical and subtropical savannas and open woodlands, which generally lack dramatic topography and have diffuse bioclimatic boundaries, we refined the models by clipping the climatically derived distribution models as follows. We first removed from the climate envelope islands and built up areas of cities and towns where no BTF have been recorded. We then brought in fine-scale vegetation mapping and land-use data in order to achieve a realised species distribution approaching BTF's area of occupancy. To do this, we determined the primary Regional Ecosystem (RE, as defined in "description" in the Regional Ecosystem Description Database; Sattler and Williams 1999, Queensland Government 2014k) designation underlying each BTF record within the suitable climate modelled areas. This step in the modelling process is important because fine-scale species-habitat associations are necessary in deriving actual area of occupancy of a species (Rondinini et al. 2006) and BTF are known to use clumped resources or use resources in a clumped fashion (Isles 2007a). We added in the RE classification in the post-model processing because RE classifications are presented at a much finer scale than the 250 m grid squares used in the bioclimatic modelling process and to upscale REs to 250 m would have resulted in loss of fine-scale resolution. Primary REs with three or fewer (unrounded) presence records were ignored as being unlikely to be favourable habitat and areas listed as water or cleared (non-remnant) were also ignored. There were 23 REs within the area that had a bioclimatic suitability of ≥0.5 for the two decimal place model and 21 REs within the area of ≥0.5 suitability for the three decimal place model. Recognising that black-throated finches may sometimes use cleared areas in proximity to suitable habitat (i.e. suitable REs), we buffered suitable REs to a distance of 1118 m. This distance was chosen as an average of the maximum distances travelled by 15 radio-tracked BTF on the Townsville Plain (Rechetelo et al. submitted). To address the tension between omission and commission errors (Rondinini et al. 2006), omission error rates were calculated by checking each of the models against recent records, which were defined as being from the year 2000 or later (2312 records). Although Rondinini et al. (Rondinini et al. 2006) state that it is impossible to quantify omission and commission error rates for opportunistically collected records (as many of the BTF records were), examining the spatial locations of BTF records and background points relative to the two models is useful, as it suggests where errors might exist. Commission error rates were estimated by checking each of the models against the 99064 background points. Using these processes we chose a final habitat model based on bioclimatic and specific mapped Regional Ecosystems, buffered to 1118 m referred to as the "habitat model". Because the Regional Ecosystem classification does not indicate pre-clearing vegetation type, we used broad vegetation group classification (BVG; Queensland Government 2014e), overlaid with non-remnants in the Regional Ecosystem classification, to determine potential cleared extents available for rehabilitation as offsets. We recognise that BVG descriptions lack the detail of RE descriptions, but this was the most systematic way to assess potential offset areas.

Resource extractive and exploratory industries extents

The BTF range overlaps with the Galilee Basin coal measure (the 'Galilee Basin'), an approximately 500 km long thermal coal deposit running roughly north-south, in central QLD (Figure 1; (Queensland Government 2011)). However, we discuss resource extraction and exploration more generally in this paper.

In QLD there are a number of extractive (mining) permit and lease types, governed by the *Mineral Resources Act 1989* and the *Petroleum and Gas (Production and Safety) Act 2004* (Queensland Government 1989, 2004). These range from permits that allow exploration activities to occur in a given area, up to permits for resource extraction. Exploration permits are no guarantee of future realised resource extraction, but do require that on-ground activities such as a drilling program are conducted in a timely fashion as detailed in a work program to be submitted to the Department of Natural Resources and Mines (Queensland Government 2014d). From Queensland Spatial Catalogue Data (Queensland Government 2015), we accumulated granted extractive and exploratory industry tenures (Exploration Permits for Coal (EPC), Exploration Permits for Geothermal (EPG), Exploration Permits for Mineral (EPM), Exploration Permits for Petroleum (EPP), Mining Claim (MC), Mineral Development Licence (MDL), Mining Lease (ML), Petroleum Lease (PL), Petroleum Survey Licence (PSL)) for QLD that were extant as of 7 September 2015. We calculated the proportion converted from one permit or lease type to another because areas subject to exploration permits have a lower chance of resource realisation than areas covered by extraction leases (e.g. MLs). We examined proportions of each tenure type in relation to modelled BTF habitat in order to gain a realistic understanding of potential threats to BTF. In parts of the southern and central Galilee Basin, some MLs and MDLs have been granted over exploration permits and proponents have submitted detailed plans of proposed impact areas (Hancock Prospecting Pty Ltd 2010a, Macmines Australia Pty Ltd 2012, SGCP 2012b, a, Adani Mining Pty Ltd 2013, Queensland Government 2013, 2015). To calculate the conversion rates from lower likelihood (e.g. EPC) to higher likelihood (e.g. ML) tenure, we included some expired exploration leases (i.e. expired before 7 September 2015; (Queensland Government 2015)) because these underpin the subsequent detailed plans referred to above. Impacts resulting from these developments are likely to include broadscale clearing and conversion to large open-cut coal mines in a roughly north-south orientation, with extensive areas of underground mining to the west of this (see Table 3). Plans also include clearing for infrastructure such as airports, railways and accommodation areas and underground mined areas are likely to be subject to subsidence (e.g. Macmines Australia Pty Ltd 2012, SGCP 2012b, Adani Mining Pty Ltd 2013). We mapped the planned extents of these developments using geo-rectified images in ArcGIS 10.2 and compared these extents to tenure maps available from (Queensland Government 2015). We overlaid the BTF habitat model with this resource tenure information to determine the overlap of these potential land uses with BTF habitat. We also overlaid the habitat model on the protected area (National Park) estate within QLD (Queensland Government 2014h), to determine its current protected extent.

We used these same resource tenure data in relation to the Galilee Basin Offset Strategy (GBOS; EHP 2013). The GBOS identifies three priorities that make up a strategic footprint within the northern Brigalow Belt and Desert Uplands bioregions: 1) high conservation value areas; 2) key north-south and east-west corridors that link to adjacent bioregions; and 3) areas with potential for rehabilitation, that is, for offsetting. We refer to these subsequently as Priority 1, 2 and 3 areas.

Results

The Maxent climate envelope models had high performances with AUCs of 0.95 and 0.97 for the two and three decimal place models, respectively. Both of the models had very low omission rates with 99.0% (2288/2312) of the recent records falling within the two decimal place model, threshold (i.e. climate suitability ≥0.5), while 94.7% (2189/2312) of the recent records fell within the three decimal place model.

When unsuitable areas such as islands and built up urban centre were removed, and the suitable RE classifications with a 1118 m buffer were added to the climate models, model omission rates were still very low, with 98.4% (2276/2312) of the recent records falling within the two decimal place model threshold (i.e. climate suitability ≥0.5), and 96.6% (2233/2312) of the recent records falling within the three decimal place model.

Although absences for mobile organisms are particularly difficult to ascertain (Elith et al. 2006), more than double the number of background points fell within the two decimal point plus RE model (1919) than the three decimal point plus RE model (976 background points) (Figure 2). Importantly, large areas of the two decimal point model had many background points and no BTF records, suggesting potential commission errors in this region. Furthermore, the total area of the two decimal point model was 50,025 km^2, while for the three decimal point model it was 15,563 km^2. For a gain in area of over three times, the two decimal point model loses 43 post-2000 omission records of BTF, an improvement of 1.9% over the three decimal point model. Thus, because the three decimal place model has no obvious areas with many background points suggesting commission error, and because of the vast geographical area required to remove a small percentage of omission errors, we have adopted the three decimal place model as the most realistic model of the area of occupancy for BTF. This is the "habitat model" hereafter.

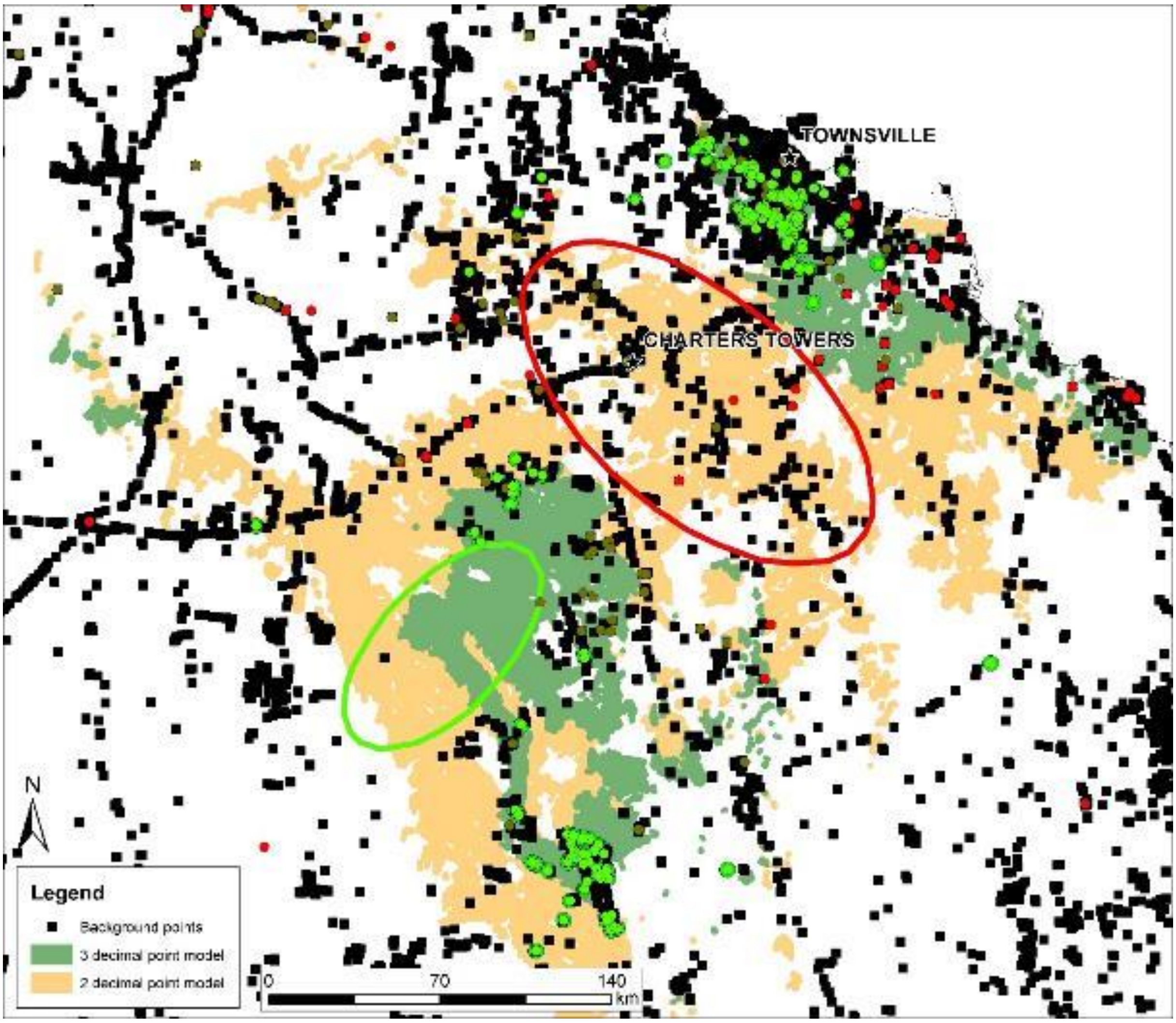

Fig. 2. Map showing areas of potential commission and omission errors in the two decimal place and three decimal place models. The red elipse enclosing the city of Charters Towers shows a large area of potential commission errors in the two decimal point model, with 282 target-group background points (i.e. bird records used for the background), with no records of BTF. The green elipse shows a very poorly sampled area. Green, brown and red dots are BTF records using the same schema as in Figure 1.

Total areas of the non-buffered REs were 6,821 km^2. A breakdown of the component favourable RE extents is given in Table 1, along with the numbers of post-2000 BTF records within each RE. Two Townsville Plains REs (11.3.30: *Eucalyptus crebra*, *Corymbia dallachiana* woodland on alluvial plains and RE11.3.35: *E. platyphylla*, *C. clarksoniana* woodland on alluvial plains (Queensland Government 2014k)) contained 74.5% of the post-2000 BTF records. In the less sampled western part of BTF range the dominant two BTF REs (10.5.5a: a variant of *E. melanophloia* open woodland on sand plains and 10.3.28a: a variant of *E. melanophloia* or *E. crebra* open woodland on sandy alluvial fans (Queensland Government 2014k)) accounted for 4.2% of the post-2000 records.

Of the habitat model, 43.1% (6,703 km^2) remains outside the areas of granted exploration leases or extractive tenure (Figure 3). Protected areas (National Parks) cover 1.5% (235 km^2) of the habitat model and nine (0.4%) out of 2312 post-2000 BTF records, and each of these records is from the Townsville Plains IBRA subregion. Of the suitable REs within the habitat model, 38.4% (2,618 km^2) remains outside granted extractive or exploratory industry interests (Table 2). Of EPCs in the Galilee Basin that have been converted to MDLs or MLs, and thus have a higher probability of going ahead as mines, and that have also submitted development plans with infrastructure and mine footprints, 42.5% of the combined original exploration area is likely to be impacted if the mines go ahead as planned (Table 3).

Table 1. Areas of modelled favourable Regional Ecosystems (RE; see Methods, *Modelling protocol*) within the three decimal point bioclimatic model, areas currently mapped as remnant, and areas not covered by extant mining or exploration leases. The number of post-2000 BTF records for each RE are given and IBRA subregions (Environment Australia 2000) are shown. Some REs have no post-2000 BTF records but are included as favourable BTF REs because there were >3 older BTF records from those REs. The habitat model referred to is the 1118 m buffered REs.

RE	Total available (km^2)	Area with no mining interest (km^2)	% no mining interest	Number of post-2000 BTF records	IBRA subregion
7.12.24a	4	4	100.0	0	Townsville Plains
7.12.65b	1	1	100.0	0	Townsville Plains
9.8.1a	113	10	9.3	0	Cape-Campaspe Plains, Undara - Toomba Basalts
9.12.1a	4	4	100.0	0	Broken River
10.3.6a	840	101	12.0	1	Cape-Campaspe Plains
10.3.28a	851	36	4.3	25	Alice Tableland, Cape Campaspe Plains
10.4.5	85	0	0.0	9	Alice Tableland, Cape Campaspe Plains
10.5.1a	134	3	2.5	2	Alice Tableland, Cape Campaspe Plains
10.5.5a	3030	501	16.5	36	Cape-Campaspe Plains
10.7.11a	189	33	17.2	1	Alice Tableland, Cape Campaspe Plains
11.3.12	214	207	97.0	169	Townsville Plains
11.3.25b	168	127	75.4	90	Townsville Plains
11.3.27	8	6	78.2	3	Townsville Plains
11.3.30	349	268	76.9	534	Townsville Plains
11.3.31	82	73	89.1	14	Townsville Plains
11.3.35	586	491	83.8	527	Townsville Plains
11.3.35a	195	186	95.4	4	Townsville Plains
11.11.9	203	187	92.0	0	Cape River Hills
11.11.15	222	187	84.1	0	Beucazon Hills
11.12.1	365	177	48.6	0	Townsville Plains, Bogie River Hills, Beucazon Hills, Belyando Downs
11.12.9	140	136	96.8	8	Townsville Plains
TOTAL REs	6,821	2,618	38.4	1,423	
HABITAT MODEL	15,563	6,703	43.1	2,233	

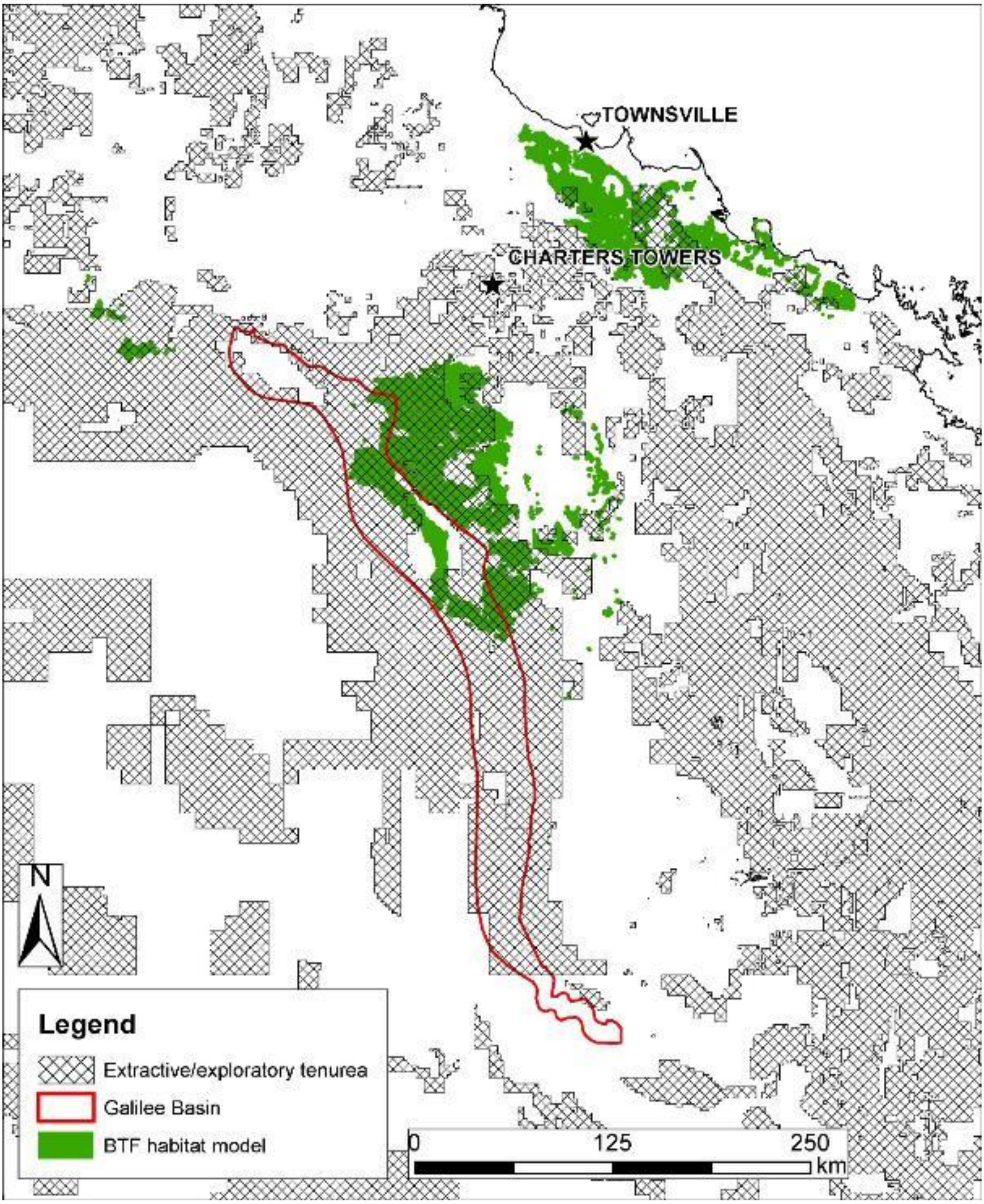

Fig. 3. Modelled BTF habitat and extant extractive/exploratory tenures. The BTF habitat model is the favourable BTF REs which are within the area of ≥0.5 bioclimatic threshold climate envelope, buffered to 1118 m.

Table 2. Total areas for mining tenures and protected areas (National Parks) within the habitat model area. Areas are incongruent with figures presented in Table 1 because some areas have more than one mining tenure over them and because of rounding. Habitat model extents as well as actual favourable RE (from Table 1) extents within NPs are also given. Tenures: Exploration Permits for Coal (EPC), Exploration Permits for Geothermal (EPG), Exploration Permits for Mineral (EPM), Exploration Permits for Petroleum (EPP), Mining Claim (MC), Mineral Development Licence (MDL), Mining Lease (ML), Petroleum Lease (PL), Petroleum Survey Licence (PSL).

	Habitat Model
Tenure	Total area (km^2)
EPC	6,756
EPM	1,044
EPP	5,194
MC	0
MDL	11
ML	298
EPG	463
National Park REs	36
National Park (buffered)	235
Total habitat model	15,563

Table 3. Total areas for EPCs and MDLs from the southern and central Galilee Basin. Alpha and Kevin's Corner are grouped together because they shared portions of EPC1210. All areas were calculated from (Queensland Government 2015) and documents referenced in the Source column using ArcGIS 10.1 geo-rectified, low resolution imagery from source documents and creating polygons around affected areas. All areas are given in hectares (Ha). Area affected is the measured impact area within the relevant lease or tenement, broken down into: OC (open cut mine); UG (underground mine); "Other" refers primarily to infrastructure such as buildings and accommodation, waste dumps, sediment dams and airports, but for one project may include an underground mine section (compare Figures 2 and 3 in Waratah Coal 2011). EPC = Exploration Permits for Coal, MDL = Mineral Development Licence

Proponent	Lease/Tenement	Area (Ha)	Area affected (Ha)	OC	UG	Other	% affected	Source
Adani	EPC1080 (east portion only)	18714	32112[ǂ]	17424	9507	5181	71.8	(Adani Mining Pty Ltd 2013)
	EPC1690	26016						
Alpha/Kevin's Corner	EPC1210	36818	50649	12969	18238	19441	49.7	(Hancock Prospecting Pty Ltd 2010a, Hancock Galilee Pty Ltd 2011, Queensland Government 2013)
	MDL285	33682						
	MDL333	31480						
China First	EPC1040	75674	53881	7168	27630	19084	38.6	(Waratah Coal 2011)
	EPC1079	63863						
China Stone	EPC987 (south portion only)	20066	16787	3589	7404	5795	83.7	(Macmines Australia Pty Ltd 2012)
South Galilee	EPC1049	89523	14823	3347	6171	5305	16.6	(SGCP 2012b, a)
TOTAL		395836	168252	44497	68951	54805	42.5	

[ǂ]Does not include approximately 2929 ha industrial area, airport and accommodation village that lie outside extents of EPC1080 and 1690.

For post 2000 BTF records that are beyond the Townsville Plains (and thus likely to be more impacted by mining tenures), 125 of 140 records (89%) are from or within 1118 m of BVG 17b (woodlands to open-woodlands dominated by *Eucalyptus melanophloia* (or *E. shirleyi*) on sand plains and footslopes of hills and ranges). The cleared extent of BVG 17b within the three decimal place bioclimatic model threshold is 279 km^2. Of the priority areas identified in the GBOS (EHP 2013), overall 40.0% (18,104 of 45,258 km^2) falls outside areas with overlying resource tenure: 49.0% of Priority 1; 24.6 % of Priority 2; and 39.6% of Priority 3.

Discussion

The entire habitat of widespread species is rarely threatened by singular events. Rather, small percentage habitat losses, fragmentation and degradation create cumulative impacts resulting in "death by a thousand cuts" (Laurance 2010). Responsibility for the survival of widespread species may be difficult to define and does not usually fall into the hands of one proponent. The regulatory framework protecting threatened species may be similarly evasive in terms of assigning responsibility. Consequently, decline and extinction of once-widespread species has occurred through multiple factors acting in concert (Brook et al. 2008).

Our model of BTF habitat shows that 56.9% of the remaining suitable habitat falls within granted, extant resource extraction or exploration tenures (Table 1). Therefore, insufficient BTF habitat exists to secure enough land to offset all the potential extraction or exploration developments. Given that the BTF has lost 80% of its historic range, losing 56.9% of the remaining habitat would be a serious threat to the species' persistence. It is unlikely that all of the extraction or exploration tenure areas will be developed as mines, but for areas with detailed mine plans, 42.5% of the original lease area is planned to be developed. Furthermore, over 80% of the area of favoured REs for BTF along the eastern edge of the Desert Uplands bioregion is under resource extraction or exploration tenures (IBRA subregions: Alice Tableland, Cape Campaspe Plains, Cape River Hills, Undara – Toomba Basalts; **Table 1**), suggesting that if approximately 42.5% of lease areas are developed, then around 50% of this stronghold is likely to be lost to mining activities. Furthermore, there is a danger that multiple exploratory activities, separate from current planned mines would result in fragmentation, habitat loss and degradation without requiring offsetting, because impacts may be perceived to be insignificant and thus not trigger further investigation.

The Galilee Basin Offset Strategy (EHP 2013) provides guidance for biodiversity offset planning for the northern Brigalow Belt and Desert Uplands bioregions, which encompass most of the BTF's remaining range. Under the strategy, offsets may be established in degraded or cleared areas that can be improved or rehabilitated in order to actually offset biodiversity losses (EHP 2013, p. 21). However, most of the eastern part of the Galilee Basin is held under coal exploration tenure by a number of companies (EHP 2013, Queensland Government 2015) and given that approximately 57% of the modelled BTF habitat could be explored and/or developed for mining, it is technically impossible to apply the current offset arrangements and achieve no net loss of BTF. Little of this key region is available for rehabilitation to offset BTF habitat loss: within the Brigalow Belt, Desert Uplands and Einasleigh Uplands, which collectively provided over 99% of post-2000 BTF records, 43.7% of the bioclimatic suitability ≥0.5 area is non-remnant (cleared). The total area of non-remnant land is considerably less than the area under extractive or exploratory tenures (Tables 1& 2), so there is a deficit of land that could be rehabilitated for BTF habitat offsets. Furthermore, cleared areas of formerly favourable habitat such as open woodlands dominated by *Eucalyptus melanophloia* (BVG17b) are even more limited. One recently approved mine (Australian Government 2014a) alone will impact approximately 97 km^2 of BTF habitat (GHD 2014a). Therefore, if cleared habitat is to be rehabilitated for offsetting purposes to the Federally required (Australian Government 2014a) extent of approximately 309 km^2, then around 28% of the cleared BVG 17b, which is the main favourable habitat impacted, would be required as offsets for this mine alone. Within recorded movement distances of BTF (16 km; Rechetelo, unpublished data, 2014) of this mine's boundary there is less than 43 km^2 of non-remnant BVG 17b, meaning close proximity offsetting is likely to be impossible (Morris et al. 2006, Australian Government 2012, Pilgrim et al. 2013). Furthermore, neighbouring applications for additional MDLs and MLs totalling at least 1047 km^2, are in place (Waratah Coal Pty Ltd 2014, Queensland Government 2015) further limiting the scope for local offsets.

Another important issue relates to the time lag for restoration to occur (Suding 2011). Nowhere within the BTF's range has intentional forward planning occurred to mitigate against time lags (e.g. GHD 2014a), nor is it a requirement under the Galilee Basin Offset Strategy (EHP 2013). Rather, the purchase or management of offsets usually begins after development commences (e.g. EPBC Approval Decision 2008/4648 2012). This strategy can only result in a net loss of habitat or environmental values (Bekessy et al. 2010, Maron et al. 2012) because it is essential for habitat to be continuously available for persistence of the species; offsets must be created before the activity that they seek to offset is undertaken (Gibbons and Lindenmayer 2007, Maron et al. 2012). To use specific examples from one mine, rehabilitation activities listed in ecofund (2013) and GHD (2014a) are likely to take many years to develop into the high quality habitat they are intended to offset. Hollow-bearing trees, for example, which may be used as nest sites for BTF are likely to take much longer than 30 years to develop (Bedward et al. 2009). Furthermore, to our knowledge, restoration has not been attempted for BTF habitat in any context. In other systems, restoration of highly degraded habitat often leads to a different ecological community than that which previously existed (e.g. Buckney and Morrison 1992, Wilkins et al. 2003, Lindenmayer et al. 2012). In addition, much of the former BTF range is invaded by buffel grass (*Cenchrus ciliaris*), which is widely favoured by graziers and is highly invasive (Franks 2002, Grice et al. 2012, Marshall et al. 2012) and has never been successfully controlled on a large scale. Where clearing has not occurred, impacts such as grazing are more easily mitigated and thus grazing land managed for BTF could potentially be used as offsets (Maron et al. 2012). However, it is not possible to assess this potential aspect of offsetting because specific details are omitted in the Environmental Offset Package (ecofund 2013; pp. 29-43) and also omitted for an adjacent mine proposal where there is potential overlap of mine developments and offsets (Macmines Australia Pty Ltd 2012, Hansen Bailey 2015).

In the Galilee Basin, other threatened species, such as the yakka skink (*Egernia rugosa*), and communities such as RE 10.9.3a (a *Eucalyptus cambageana* woodland), are also likely to be impacted by exploratory or extractive industries to an extent that is difficult or impossible to offset. For example, the entire extent of the *Eucalyptus cambageana* woodland community is within areas of extractive or exploration tenure. For many species and communities, the land available for offsets is limited; so offsets may come in the form of research funding. Although useful for understanding the ecology of the species as a basis for improved conservation, research funding offsets have little direct benefit in actually conserving habitat or protecting the population (McKenney and Kiesecker 2010). Approximately 60% of the area designated by the strategic footprint in the GBOS (EHP 2013) as potential offsets against loss of biodiversity is itself covered by resource extraction or exploration tenures. Priority 3 areas occupy a greater extent than Priority 1 areas (8,075 km^2 vs 7,867 km^2) as they must to adequately offset areas that are in better condition (Bekessy et al. 2010, Maron et al. 2012). However, the extent of Priority 1 areas under exploratory or extractive tenure is 8,196 km^2, whereas extent of Priority 3 areas (potentially to be used as offsets) *not* under exploratory or extractive tenure is 8,075 km^2, slightly less than the area it is supposed to offset. Thus, the areas actually available for offsets is less than the area that would be required for a 1:1 offset ratio.

Because BTF are primarily ground feeders dependent on seeds accessed on relatively open ground (Higgins et al. 2006b), they are vulnerable to habitat alteration by invasive species such as grader grass (*Themeda quadrivalvis*) and shrubby stylo (*Stylosanthes scabra*) (Rechetelo, unpublished data, 2014). Fragmentation is likely to make movement across the landscape more difficult and reduce population viability (BTFRT 2007b, Garnett et al. 2011b). Our analysis is likely to underestimate the impacts of the mining developments discussed because we do not account for the impacts of fragmentation, which increase the likelihood of incursion by invasive species, and could change fire regimes, leading to overall lower suitability (Saunders et al. 1991a, Brook et al. 2008, Wilson et al. 2009). Fragmentation would result, for example, from the railway corridors that are planned to service extractive industries in the Galilee Basin; these have not been considered in this paper, but would be long (approx. 495 km: Hancock Prospecting Pty Ltd 2010b, e.g. 189 km x 95 m: Adani Mining Pty Ltd 2014) fragmentation barriers. Also not considered to this point in this paper is potential habitat fragmentation as a result of gas drilling, which has an inherently high edge to footprint ratio because of well spacing and access roads (Queensland Government 2014a). This is being undertaken at the southern edge of the BTF's current range.

Our results show that adequate provision of offsets to provide protection for BTF is likely to be a difficult proposition in the stronghold area of the eastern Desert Uplands. Protection of remnant high value habitat should not be considered as offsetting as this will result in a net loss of suitable habitat (Bekessy et al. 2010), and protection of offsets if they are restored from cleared or degraded land is likely to be problematic for a number of reasons. First, conditions on approvals (e.g. Eco Logical Australia 2012b) require offset areas to be 'legally secured' for at least the duration of the impact (Australian Government 2012, Queensland Government 2014i, b). There may be a recommendation of 'in perpetuity' protection (EHP 2013), but the security of offsets is questionable because they may be revoked (Queensland Government 2014c), Nature Refuges may be developed for mining (Waratah Coal 2011), and even for National Parks, there is currently a designated financial offset ratio (10:1) that may, potentially, be proponent-driven (Queensland Government 2014j) and thus potentially not available for public scrutiny (Hansen Bailey 2015). This policy framework undermines the prospects for secure offset protection for BTF. Second, the ecological requirements of BTF are poorly understood. Although preferred habitats are generally known (see model), there is no established means of rehabilitating heavily degraded or cleared land as BTF habitat. This further undermines the prospects for confidently using offsets as a protection mechanism. Third, timeframes given in offset documents such as 'for the duration of the impact', or 'until 2073' (EPBC Approval Decision 2008/4648 2012), are likely to be insufficient as a long-term protection mechanism and provide little guarantee of offset success (Pilgrim and Bennun 2014).

Other land use factors threaten the persistence of the BTF, particularly in other stronghold areas such as the Townsville Plains, where only 34% of the sub-region is planned for resource extraction. The BTF population in this area occurs on the fringe of Australia's largest tropical city and is under threat from ongoing urban expansion, weed invasion, habitat fragmentation and possibly invasive animals. The human population of the Townsville Local Government Area is forecast to expand 122-134% over 2011 levels by 2021, while the broader Townsville region is forecast to expand by 118-128% over the same time frame (Queensland Government 2014f, g). There is no reliable BTF population estimate and the large number of records for this region (Figure 1, Table 1) does not equate to a large, stable or secure population but rather proximity to an urban centre with many bird observers, including annual water hole counts since 2004 (BTFRT, unpublished data).

While the plight of the BTF is being considered under federal threatened species legislation, we show here that current mitigation strategies are unlikely to be sufficient to prevent further severe decline. To make a genuine effort to avoid net loss of a species facing habitat loss, stricter protocols such as those proposed by (Bos et al. 2014) need to be in place. Primarily in the context of groundwater, the Independent Expert Scientific Committee on Coal Seam Gas and Large Coal Mining Development (IESC 2013) considered that, given the scale of proposed developments within the Galilee Basin, information on cumulative impacts should be commensurate with the scale of all proposed developments. The same holistic approach should be taken when considering development approvals and conditions so that overall risks to a species can be fully evaluated. The area available for offsets is likely to be diminished when many developments occur in one region, and likely to be uncoordinated in the absence of a statutory overarching strategic plan.

Our approach has looked broadly at scope for establishing offsets for BTF in central QLD in the face of planned and prospective broadscale landscape change. Our findings show that there is insufficient existing habitat for BTF that could be used to offset the mining developments planned within its range. Furthermore, insufficient land exists that could be restored to increase the area of habitat. However, even if sufficient land did exist it is unlikely that habitat of sufficient quality could be created within the timeframe required.

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