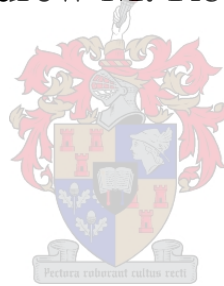


Avian assemblages of invasive Australian *Acacia* thickets in the Western Cape

by

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Thesis presented in partial fulfillment of the requirements for the degree of Masters of Science in the Faculty of Science, Department of Botany and Zoology, at Stellenbosch University

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Declaration

I, the undersigned, hereby declare that the entirety of the work contained therein is my own original work, that I am the sole author thereof, that reproduction and publication thereof by Stellenbosch University will not infringe any third party rights and that I have not previously in its entirety or in part submitted it for obtaining any qualification.

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Abstract

Human-modified habitats form increasingly large components of landscapes, threatening biodiversity and creating challenges for conservation. In some cases altered habitats form entirely novel ecosystems that may support new combinations of species and species abundances, and create habitat space in otherwise transformed landscapes. In the Western Cape of South Africa, woody invasive species contribute to landscape-level habitat transformation and form novel ecosystems. Invasive Australian *Acacia* species are especially problematic in lowland areas where they create dense thickets and substantially transform both biotic communities and abiotic processes. Despite the prominence of *Acacia* thickets across the Western Cape, their ability to support native fauna is not well understood and the objective of this study was to assess the significance of *Acacia* thickets as habitat for the region's avifauna. Birds were surveyed in *Acacia* thickets in the south-western Western Cape in three seasons to examine species richness, abundance and functional abundance. Furthermore, I examined the extent to which differences in patch-level vegetation structure alter bird communities. Between survey sites and seasons, significant variation was observed in assemblage richness, density, median body size and biomass. Variation in vegetation density, stem density, mean vegetation height and total canopy cover best explained variation in bird assemblages. Eighty species were estimated to utilize *Acacia* thickets and assemblages had a mean density of 7.78 birds per ha. The most abundant feeding guilds were the mixed feeders and insectivores. The median body size observed was 15.2 g and the body size frequency distribution of all species in *Acacia* spanned a similar range compared to the body size frequency distribution for the species list for the entire Western Cape. The mean biomass of bird communities was 0.224 kg per ha. Using data on bird density and biomass, *Acacia* thickets across the Fynbos Biome support and estimated average of over 21 million birds with a combined biomass of over 600 thousand kg. This study found that *Acacia* thickets in the Western Cape support a subset of the region's birds with the most abundant species being small mixed feeders, which are also frequently urban-adapted. Compared with other habitat types, *Acacia* support bird assemblages with moderate species richness and density. This study shows that *Acacia* thickets, as a novel habitat, provide a significant amount of habitat space in a highly transformed landscape and highlights the need for comprehensive evaluation of altered habitats before assumptions are made about their ecological value.

Opsomming

Getransformeerde habitatte maak vermeerderend groot deel uit van die omgewing, dit bedreig biodiversiteit en skep groter uitdagings vir bewaring. In sommige gevalle vorm hierdie getransformeerde habitatte geheel nuwe ekosisteme wat moontlik nuwe kombinasies van spesies en spesie volopheid kan onderhou. Verder skep nuwe ekosisteme habitat spasie in anders veranderde landskappe. In die Wes-Kaap van Suid-Afrika dra die Australiese *Acacia* indringer spesies is veral problematies in laagliggende areas, aangesien dit digte ruigtes vorm, asook beide die biotiese gemeenskappe en die abiotiese prosesse aansienlik transformeer. Ten spyte daarvan dat daar volop *Acacia* ruigtes in die WesKaap is, word min verstaan van hul vermoë om inheemse fauna te onderhou. Die hoofdoel van hierdie studie was om die belang van *Acacia* ruigtes as habitat vir die area se voëllewe te bepaal. Voël-opnames in die suidwestelike dele van die Wes-Kaap is gedoen in *Acacia* ruigtes oor drie seisoene, om spesierykheid, volopheid en funksionele volopheid te ondersoek. Verder is die mate waartoe verskille in die plote van die plantegroei struktuur, die voëlgemeenskappe verander, geondersoek. Daar was aansienlike variasie waargeneem in die spesiesamestelling rykheid, voorkoms digtheid, mediaan liggaamsgrootte en biomassa van die voëls tussen die onderskeie voëlopnaam plote en die seisoene. Die variasie in plantegroei digtheid, stam digtheid, mediaan plantegroeihoogte en totale kroonbedekking verduidelik hierdie variasie in spesiesamestelling die beste. Tagtig voëlspesies *Acacia* ruigtes benut en die populasiesamestelling het 'n gemiddelde digtheid van 7,78 voëls per ha. Die mees algemene voel-voeding-guldes was die gemengde-voedsel-vreters en insekvreters. Die median liggaamsgrootte waargeneem was 15,2 g en die liggaamsgrootte frekwensieverspreiding van alle spesies in *Acacia* ruigtes is ooreenkomstig met die liggaamsgrootte frekwensieverspreiding vir die spesielys vir die hele Wes-Kaap. Die gemiddelde biomassa van voel gemeenskappe was 0.224 kg per ha. *Acacia* ruigtes oor die fynbosbiom wat 'n geskatte gemiddelde van meer as 21 miljoen voëls ondersteun, met 'n gesamentlike biomassa van meer as 600 duisend kg. Hierdie studie het bevind dat *Acacia* ruigtes in die Wes-Kaap 'n onderafdeling van die streek se voëls ondersteun, met die mees algemene spesies as die klein gemengde-voedsel-vreters, wat ook dikwels stedelik aangepas is. In vergelyking met ander habitattipes ondersteun *Acacia* ruigtes voel samestellings met matige spesierykheid en digtheid. Hierdie studie toon dat die *Acacia* ruigtes, as 'n nuwe habitat, 'n beduidende hoeveelheid habitat ruimte in 'n hoogs getransformeerde omgewing skep en beklemtoon die behoefte aan 'n omvattende evaluering van veranderde habitatte, voor aannames gemaak word oor hul ekologiese waarde.

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Chapter 1 Introduction

Invasive *Acacia* thickets as novel ecosystems: Impacts and services

Habitat transformation from anthropogenic sources is the leading cause of global biodiversity loss (Bolger *et al.* 1991; Pimm *et al.* 1995; Chown, 2010). Agriculture, urbanization, and invasive species have altered and degraded indigenous ecosystems (Tilman *et al.* 2001; Rouget *et al.* , 2003; Buchanan *et al.* 2009) and reduced native community richness and abundance, resulting in homogenized communities dominated by generalist species (Rooney *et al.* 2007; Clavel *et al.* 2011). Conserving native species within modified landscapes becomes increasingly difficult as habitat fragmentation, isolation and exploitation increase pressure on indigenous species populations (Hansen and Rotella, 2002; Donnelly and Marzluff, 2004; Buchanan *et al.* 2009; Harrison and Bruna, 2009; Hodgson *et al.* 2011). Furthermore, global climate change is predicted to shift species ranges, changing the effectiveness of existing reserves to support native species (Erasmus *et al.* 2002; Midgley *et al.* 2002; Coetzee *et al.* 2009) and complicating species response to habitat transformation (Hockey *et al.* 2011). Species persistence within these altered and changing landscapes will depend in part on their ability to disperse through and utilize available resources (Winker *et al.* 1995; Fahrig, 2003, 2007). As native habitat diminishes, the relative value of marginal and landscape-matrix habitats becomes more important for species conservation (Prevedello *et al.* 2010; Edwards *et al.* 2011).

Invasive alien species are among one of the most important drivers of anthropogenic habitat transformation (MacDougall and Turkington, 2005; Pyšek and Richardson, 2010). They degrade native ecosystems by altering biotic and abiotic ecosystem structure and function (Mack and Antonio, 1998; Richardson and van Wilgen, 2004; van Wilgen *et al.* 2008; Vilà *et al.* 2011) and establish in both modified and natural areas (Rouget *et al.* , 2003). The introduction of alien and invasive species is increasing globally, driving species extinctions through novel interactions (including predation or mesopredator release), habitat degradation and fragmentation (Gaston *et al.* 2003; Blackburn *et al.* 2004; Butchart *et al.* 2010). For introduced species to become invasive they must become naturalized, reproduce and disperse without the aid of humans (Blackburn *et al.* 2011). The consequent impact of an invasive species on native communities depends on the traits of the species invading (Yelenik *et al.* 2004; Ricciardi and Cohen, 2007; Gaertner *et al.* 2009), the composition of the native community (Rodriguez, 2006; Vilà *et al.* 2011) and the landscape context of the invasion

(Yelenik *et al.* 2004; van Wilgen *et al.* 2008; Dures and Cumming, 2010). The most problematic invasive species are those which alter biotic structure and abiotic processes in a downward degradation pathway and in doing so prevent autogenic recovery of native communities (King and Hobbs, 2006; Brooks *et al.* 2010). Some invasive species change the timing or severity of disturbance regimes and can create positive feedback cycles which maintain altered, invaded ecosystem states (Brooks *et al.* 2010; Le Maitre *et al.* 2011).

In highly altered habitats, alien or invasive species can establish novel ecosystems that have an anthropogenic source, no contemporary analogue, new biotic and abiotic conditions, as well as new combinations and abundances of species (Milton, 2003; Hobbs *et al.* 2006). In highly transformed and degraded areas where native vegetation is unlikely to recover, novel ecosystems can provide habitat space that supports diverse and abundant native assemblages (Lugo and Helmer, 2004; Seastedt *et al.* 2008; Quine and Humphrey, 2010; Chown and McGeoch, 2011). However, this potentially beneficial service is often overshadowed by negative perceptions of altered, invaded and otherwise degraded habitats (Milton, 2003; Davis *et al.* 2011). Although novel ecosystems cannot support full suites of native species, the relative value of altered habitats in the landscape will increase as undisturbed habitats are increasingly lost, fragmented and isolated (Quine and Humphrey, 2010; Edwards *et al.* 2011). The formation of novel habitats is certain to have both positive and negative impacts on native communities (Williams and Jackson, 2007; Lindenmayer *et al.* 2008; Hobbs *et al.* 2009); however, predicting the ability of novel systems to support native species is hampered by limited knowledge of how communities assemble in altered habitats and requires context specific studies (Brooks *et al.* 2010).

In South Africa, much of the Western Cape Province is invaded by woody, thicket-forming Australian *Acacia* species which are among the country's most problematic plant species due, in part, to their ability to transform native habitats (Henderson 2001). Australian acacias have been and continue to be introduced around the world for forestry, horticulture, and ecological function purposes despite a known propensity to become invasive. Among *Acacia* species introduced to South Africa, 15 species are listed as weedy or invasive (Richardson *et al.* 2011). *Acacia saligna* and *A. cyclops* were introduced and intensively planted in the Western Cape during the 19th and 20th centuries for horticultural purposes and to stabilize the sandy soils (Poynton, 2009). More recently, these species have become important for construction material, animal fodder and fuel wood for previously disadvantaged communities. As a consequence of early, extended planting efforts and their importance to humans, *A. saligna* and *A. cyclops* have become two of the most problematic invasive species in the lowland fynbos communities (Richardson and van Wilgen, 2004; Yelenik *et al.* 2004;

Richardson and Kluge, 2008). Their success is due to, in part, their ability to fix nitrogen and grow rapidly in nutrient poor soils (Musil and Midgley, 1990; Witkowski, 1991), an early age of reproduction (Gibson *et al.* 2011), successful seed dispersal by native animals (Glyphis *et al.* 1981; French and Major, 2001; Underhill and Hofmeyr, 2007), and long-lived, disturbance-triggered seed bank (Holmes and Cowling, 1997; Richardson and Kluge, 2008). *Acacia saligna* and *A. cyclops* alter abiotic conditions in invaded areas by changing soil nutrients (Musil and Midgley, 1990; Witkowski, 1991), increasing leaf litter, increasing standing biomass and changing surface moisture profiles (Yelenik *et al.* 2004; Gaertner *et al.* 2011). Additionally, *Acacia* trees form dense thickets (Figure 1) that alter vegetation communities by shading out native species and alter the return rate and intensity of fire (van Wilgen and Richardson, 1985; Holmes and Cowling, 1997; Gaertner *et al.* 2011). Altered abiotic conditions and biotic structure following *Acacia* invasion creates feedback cycles that maintain *Acacia*-dominated habitats (King and Hobbs, 2006; Le Maitre *et al.* 2011).

The impacts of novel Australian *Acacia* systems on biotic communities are not well-understood. Recent studies investigating the impact on different invertebrate assemblages found that not all groups responded in similar ways. French and Major, (2001) found that *Acacia* invasion reduced total ant abundance and modified community composition but that total richness did not change. Invasion increased the abundance of a few species, although declines in abundance were found for many more species. Samways *et al.* (1996) showed that terrestrial arthropod assemblages in *Acacia*-invaded areas had reduced functional richness and that generalist species abundance increased. Procheş *et al.* (2008) found that arthropod assemblages within *Acacia* canopies typically had higher species richness than those within the canopies of indigenous vegetation in summer, but not in winter, and assemblage abundance was always higher in indigenous assemblages. This effect is especially pronounced among herbivorous species.

The extent to which invasion by *Acacia* alter bird assemblages and site occupancy depends on the density of *Acacia* trees and landscape context in which the invasion occurs. A study by Winterbottom (1968) compiled the first list of bird species utilizing Australian *Acacia* thickets using field cards from the Cape Bird Club to catalogue species. Using frequency of occurrence on the field cards as a proxy for abundance, 86 species were recorded as utilizing *Acacia* trees, but only 11 species were considered common (common being determined by presence on at least 40% of field cards). Furthermore, *Acacia* supported more species than coastal renosterbos, ploughed land, pastures and grain fields, but supported fewer species than coastal fynbos and Strandveld. Only one species, the Fiscal Flycatcher (*Sigelus silens*), was recorded in *Acacia* trees but not in native vegetation, while 12 species recorded in

fynbos were absent from *Acacia* thickets. Additionally, Winterbottom (1968) found that *Acacia* thickets shared more species with native vegetation than habitats with mixed woody invaders (i.e. *Eucalyptus*, pines and oaks). While this study was a useful first assessment of bird communities in transformed *Acacia* habitat, it lacked the data to provide true abundance estimates. Fraser and Crowe (1990) studied bird assemblages along a gradient of *Acacia cyclops* invasion on the Cape Peninsula in Table Mountain National Park (formally Cape of Good Hope Nature Reserve) and found invasion did not significantly alter bird density, richness or biomass, but did excluded some nectarivorous species. Birds found in dense *Acacia* stands were more typical of native woodland or thicket habitats and the highest density of birds was recorded in plots with 10 and 50 % *Acacia* cover. Seven species recorded in the densest *Acacia* thickets were not present in native vegetation plots, while only the Cape Sugarbird (*Promerops cafer*) was absent from the most invaded sites. Although Fraser and Crowe (1990) were able to capture density and richness estimates, their study was limited in geographical extent preventing extrapolation of their findings to *Acacia* thickets in different landscape contexts. In this study, I expand on the work by Winterbottom (1968) by directly surveying bird density, and test whether the patterns found by Fraser and Crowe (1990) are consistent in invaded sites across landscape contexts. This study provides a first assessment of bird occupancy of this novel system at broad spatial scales allowing for a more detailed understanding of the role of *Acacia* thickets as habitat space for native animals. This work provides important insights for management of invaded habitats and conservation of birds in a transformed landscape.



Figure 1. *Acacia* invasion drastically changes native habitats. *A. saligna* can reach high densities forming dense thicket with little understory (a), while at lower densities supporting an alien grass understory and forming open savannah habitats (b). *A. cyclops* has a different growth structure and forms dense thickets with little understory and more uniform vegetation profile (c). All thicket habitats differ significantly from the native fynbos vegetation, as seen in the foreground of (d).

Community assembly in novel habitats

Anthropogenic habitat alteration and degradation are the leading threats to biodiversity, and contribute to the formation of novel systems (Milton, 2003; King and Hobbs, 2006). As undisturbed habitats are altered, understanding how communities assemble within novel systems (Luck and Korodaj, 2008; Sweeney *et al.* 2010) and to what extent native species are supported in degraded habitats have the potential to contribute to conservation efforts (Lindenmayer *et al.* 2008; Quine and Humphry, 2010; Edwards *et al.* 2011). Habitat transformation is known to alter community composition, habitat structure, habitat connectivity, and landscape heterogeneity, with consequences for community species richness, functional richness and species abundance (Crooks, 2002; Jones and Bock, 2005; O'Connor and Crowe, 2005; Mayfield *et al.* 2010). However, despite some success with novel

restoration and management techniques (Lindenmayer *et al.* 2008; Seastedt *et al.* 2008), current understanding of how communities assemble within altered habitats is not detailed enough to predict the trajectory of community assembly or to purposely guide systems toward desired states (Brooks *et al.* 2010).

Bird occupancy of altered habitats varies across both spatial and temporal scales. At local spatial scales floristic composition is an important driver of niche availability and bird species richness (Willson, 1974; Rotenberry, 1985; Champlin *et al.* 2009; Fleishman *et al.* 2009). Habitat alteration reduces resource diversity at small spatial scales, resulting in a loss of habitat-specialist birds and creating homogenized assemblages dominated by generalist species (Diamond, 1975; Easton and Martin, 1998; Dean *et al.* 2002; Devictor *et al.* 2007; Clavel *et al.* 2011). However, in some places even common species are declining and warrant monitoring and conservation efforts (Gaston and Fuller, 2007). At larger spatial scales, vegetation structure, total area and surrounding habitat become more important predictors of bird habitat occupancy (Rotenberry, 1985; Armstrong and Van Hensbergen, 1994; Davis, 2004). Across landscapes, habitat alteration increases habitat heterogeneity and may add previously absent resources which increases species richness (Fairbanks, 2004; Mitchell *et al.* 2005). For example, the expansion of agriculture, water sources and woody alien trees in the Western Cape has promoted the range expansion of species able to utilize these regionally novel habitats, with the effect of increasing regional species richness (Hockey and Midgley, 2009; Hockey *et al.* 2011).

At broader spatial scales, total habitat area, fragmentation and connectivity all contribute to the number and types of birds occupying a particular habitat. As landscapes are transformed, the arrangement, fragmentation and connectivity of habitats are altered, changing bird assemblages (Devictor and Julliard, 2008; Shanahan and Possingham, 2009). The resource pool and structure of altered habitats changes species' ability to move through modified landscapes (Winker *et al.* 1995; Fahrig, 2007; Betts *et al.* 2008). Landscape permeability depends on the quality of the matrix through which species move (Sisk *et al.* 1997; Wethered and Lawes, 2003; Fahrig, 2007; Tremblay and Clair, 2011), species-specific responses to modified environmental cues (Tremblay and Clair, 2011) and survival during dispersal (Verner, 1992). Accounting for landscape scale patterns in habitat configuration is therefore an important part of understanding why certain birds occupy certain habitats.

As community species composition changes with habitat transformation, functional richness may also change, with consequences for ecosystem functioning (O'Connor and Crowe, 2005; Foster and Robinson, 2007). Altered food resources, foraging substrates, and nesting sites change niche space available within a habitat. Occupancy of habitat specialists is

often correlated with plant species composition and structure (Deppe and Rotenberry, 2008; Champlin *et al.* 2009), thus extirpation of plant based food resources following invasion can alter habitat suitability quickly for specialist species (Fox and Hockey, 2007). In South Africa, habitat fragmentation and pine tree invasion results in the loss of floristic diversity that has lead to the exclusion of species and even entire feeding guilds (Armstrong and Van Hensbergen, 1994; Mangnall and Crowe, 2003; Fox and Hockey, 2007). In novel systems where invasive plants are structurally similar to native vegetation, bird communities were found to be subsets of those in native vegetation, and the absence of a few specialized species in altered communities was attributed to the lack of requisite plant species (Fraser and Crowe, 1990; Dean *et al.* 2002).

Quantifying bird abundance is integral to understanding how communities are structured (Wiens, 1989; Currie and Fritz, 1993), and techniques for estimating bird density have generated vast and well reviewed methods for abundance estimates and analysis (Bibby *et al.* 2000; Buckland *et al.* 2004). Density of animals within a habitat is the result of available resources, diet, species energy use, and competition (Currie and Fritz, 1993; Gaston, 1997; Clergeau *et al.* 1998; Champlin *et al.* 2009). Abundance is important for understanding population dynamics and species persistence within a landscape (Lawton, 1990; van Rensburg *et al.* 2000; Salomon *et al.* 2006; Gaston and Fuller, 2007). Of particular concern, within altered landscapes, is extinction debt, which may arise if transformed habitats support species in low abundances, such that the species populations are not viable over long time scales (Soulé *et al.* 1988; Rodewald *et al.* 2011). However, for some bird species, population stability in high quality habitat relies on the abundance of birds in non-breeding habitat. For example, non-territorial, non-breeding bird individuals have been shown to occupy habitat that is sub-optimal for breeding, and the abundance of such individuals buffers breeding populations against detrimental stochastic events (Verner, 1992; Kristan III *et al.* 2007; Penteriani *et al.* 2011). The abundance of birds within a habitat patch may reflect resource availability and quality within a site (Sergio *et al.* 2003; Ohnson, 2007; Betts *et al.* 2008). However, environmental cues in some altered habitats can attract high abundances of birds to low quality habitats (Bock and Jones, 2004; Champlin *et al.* 2009; Rodewald *et al.* 2011). For some territorial birds, optimal habitat can have lower bird densities due to increased incentives for the most fit birds to exclude conspecifics from a habitat patch (Sergio *et al.* 2003; Davis, 2004). Therefore, territoriality can lead to higher densities and higher individual turnover in lower quality habitat when sub-optimal competitors and juveniles occupy, but do not defend, patches as they wait for higher quality territories to become available (Winker *et al.* 1995; Boulinier *et al.* 2001). Higher densities can also arise due to clustering of territories

through conspecific attraction (where birds use each others' presence as an estimator of habitat quality), and can lead to bias in estimates of habitat quality regardless of actual variation of within-habitat variables (Ahlering and Faaborg, 2006). For many non-territorial birds, density has been shown to correlate positively with habitat quality (Lloyd, 2008). However, caution is required when making assumptions about habitat quality based on bird abundance as there is an increased chance of density being a misleading indicator of reproductive success at anthropologically disturbed sites (Sergio *et al.* 2003; Bock and Jones, 2004).

Another predictor of species density is body size, with larger species being rarer, or less abundant than small species (Peters and Wassenberg, 1983; Currie and Fritz, 1993; Silva *et al.* 1997). Body size reflects patterns in species energy consumption (Ricklefs *et al.* 1996; Glazier, 2008), thermoregulation (Angilletta and Dunham, 2003), predation pressure (Blackenhorn, 2000), habitat structure (Schmidt and Jensen, 2005) and niche availability (Telleria and Santos, 1995; Blouin-Demers *et al.* 2007) and geographic range size (Gaston and Blackburn, 2009). Median body size of bird communities is positively correlated with eNPP (a measure of primary productivity and growing season) as well as the regional species richness at large spatial scales (Greve *et al.* 2008; Huston and Wolverton, 2011). The positive correlation between richness and body size, in addition to the negative correlation between body size and abundance, explains how some altered habitats can support high bird biomass despite low bird richness and low resource diversity (Farina, 1997; Heikkinen *et al.* 2004; Walker, 2006; Hulme, 2007). However, habitat transformation alters habitat characteristics in context-specific ways (Mack *et al.* 2000), preventing generalizations about the relationship between habitat transformation, species richness, median body mass and density.

Aims of this study

Given the growing significance of novel ecosystems globally (Hobbs *et al.* 2006) and the extent of *Acacia* invasion in the Western Cape (Rouget *et al.* , 2003; Rebelo *et al.* 2006), this study set out to investigate bird species richness and density within *Acacia* habitat. Moreover, this work also provides insights into the functional groups of birds that might do best in these novel ecosystems. Investigations of the effects of landscape transformation generally on avian assemblages both in the Fynbos Biome and elsewhere in the country have shown that insectivores and nectarivores will decline in abundance, but seed feeders and mixed feeders (omnivores) increase (Greve, 2006). However, the extent to which groups change seems to vary with disturbance type (Armstrong and Van Hensbergen, 1994; Dean *et al.* 2002; Fox and

Hockey, 2007). Therefore, establishing the extent to which such generalities apply to the specific effects of transformation of fynbos by invasive *Acacia* species will provide insights into the relative costs and benefits to the avifauna of such landscape transformation. Specifically the aims of the study are:

1. Examine the richness, abundance and biomass of avian assemblages in areas invaded by *A. saligna* and *A. cyclops*, across the south-western region of the Western Cape to assess the extent to which they serve as habitats for birds in the region.
2. Determine which species and functional groups (as determined by body size and diet) are most abundant in *Acacia* thickets.
3. Determine whether variation in *Acacia* thickets at the landscape scale and patch scale affects assemblage richness, abundance and functional composition.
4. Provide an estimate of the total number of individuals and biomass of birds supported by all *Acacia* thickets in the Western Cape.
5. Examine the extent to which these findings inform broader principles regarding avian assemblage structure especially in novel habitats.

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Chapter 2: Avian communities within invasive alien *Acacia* thickets

Introduction

Habitat transformation from anthropogenic sources such as agriculture, urbanization and invasive species drives biodiversity loss (Pimm *et al.* 1995; Allen *et al.* 1999; Chown, 2010) by changing habitat suitability for native species. Because habitat transformation continues to reduce, fragment and isolate native habitats, the relative conservation value of some altered habitats for conserving indigenous species has increased (Edwards *et al.* 2011). The persistence of native fauna within and across modified landscapes will depend in part on their ability to utilize resources in transformed habitats (Markovchick-Nicholls *et al.* 2008; Pryke *et al.* 2010; Hodgson *et al.* 2011). However, current understanding of how species respond to habitat change is not sufficient to predict the extent to which altered habitats can support native communities (Brooks *et al.* 2010). Although protected areas remain the best option for conserving species (Gaston *et al.* 2008; Greve *et al.* 2011), they are under increasing pressure from transformation, exploitation and climate change driven by expanding human populations (Chown *et al.* 2003; Buchanan *et al.* 2009). Studies of communities within altered habitats are needed to understand the extent to which human dominated landscapes support local biodiversity (Lindenmayer *et al.* 2008; Hobbs *et al.* 2009).

Human-mediated introductions of non-native species contribute to the homogenization of global and local ecosystems (Rooney *et al.* 2007), local biodiversity loss (Richardson and van Wilgen, 2004) and the degradation of ecosystem services (van Wilgen *et al.* 2008; Gaertner *et al.* 2011). Negative perceptions about invasive species have led to the assumption that invaded habitats have little ecological value. However, recent studies suggest that some invasive species can have positive impacts on native species (Rodriguez, 2006; Schlaepfer *et al.* 2011), especially in highly transformed landscapes (Quine and Humphrey, 2010). Alien species can benefit native species through services such as pollination (Dick *et al.* 2003) and seed dispersal (Chimera and Drake, 2010), by providing food resources (Geerts and Pauw, 2009), enhancing native vegetation regeneration (Lugo and Helmer, 2004) and creating habitat space for endangered species (Walker, 2006; Sogge *et al.* 2008).

The suitability of invaded habitats for wildlife, such as birds, depends on species- and context-specific responses to invasion and this has prevented generalizations of positive or negative impacts of alien species for local avian assemblages (Rodewald, 2011). Several

studies have shown that invasive plants can play important beneficial roles in supporting native bird species and communities, especially in transformed landscapes. In the south-west United States, invasive *Tamarix* trees in medium densities support higher abundances of birds than native vegetation alone (van Riper III *et al.* 2008), and alien grasslands have formed critically important habitat for endangered habitat specialists (Jones and Bock, 2005). In plantations of exotic pines in the U.K., bird communities are supported which are as species rich as those in native Oak woodland (Quine and Humphrey, 2010). However, even in these examples, alien-dominated habitats were not more beneficial than native vegetation, but supported more species than highly transformed habitats with no vegetation. Invasive plants may support birds by providing replacement, or novel, food and structural resources, which are important drivers of habitat selection. However, care must be taken to evaluate the relative value of alien dominated habitats as altered environmental cues within them create potential for the formation of ecological traps (Rodewald *et al.* 2011).

In the Western Cape Province, South Africa, woody invasive species such as pines, eucalypts and Australian *Acacia* have become dominant features of the landscape and are a leading cause of landscape transformation in this biodiversity hotspot (Rouget *et al.* , 2003). The impacts of these trees depend on the traits of the exotic species as well characteristics of the native communities invaded (Gaertner *et al.* 2011). Alien *Acacia* species in lowland areas of the south-west Cape are especially problematic as they are widespread and cause significant habitat change (Yelenik *et al.* 2004; Rebelo *et al.* 2006). Only 3% of lowland areas is formally protected, which is insufficient to conserve the several endangered vegetation types which were once common in lowland regions (see description of study region in this study; Holmes *et al.* 2008). Furthermore, *Acacia* species cover 11% of lowland areas in dense thickets and have low densities on a further 33% (Rebelo *et al.* 2006) contributing to both patch and landscape level transformation. At the patch level, *Acacia* thickets differ significantly from indigenous vegetation in their physical structure and ecological functioning, creating an entirely novel habitat type in the Cape lowlands (Hobbs *et al.* 2006; see references in Le Maitre *et al.* 2011). Despite the high level of *Acacia* cover at the landscape scale, and the local changes that occur with invasion, animal communities occupying such habitats are not well described.

The suitability of *Acacia* thickets for bird communities at the landscape scale is particularly significant given the high mobility of birds and the importance of habitat structure in determining habitat occupancy (Willson, 1974; Sweeney *et al.* 2010). Winterbottom (1970) compiled the first species lists for birds recorded in *Acacia* thickets in the Cape. The study found fewer species in *Acacia* than in native vegetation types, but suggested that *Acacia*

supported bird communities which were more similar to those in native vegetation than to those in habitats formed by mixed woody invaders (pines, eucalyptus and oak). Furthermore, *Acacia* supported more species than other transformed habitats, such as ploughed land, pastures and grainfields. More recently, Fraser and Crowe (1990) found that *Acacia* invaded sections of the Cape of Good Hope Nature Reserve (now Table Mountain National Park) supported similar richness and abundance of birds. In contrast, Dures and Cumming (2010) found that *Acacia* presence negatively affected species richness, but the strength of this relationship varied with level of urbanization in the surrounding landscape. The current understanding of the bird assemblages which utilize *Acacia* as habitat presents mixed messages concerning the suitability of *Acacia* thickets for native species. Specifically a better understanding of bird abundance, biomass and functional groups is needed to make relevant ecological comparisons between *Acacia* communities and communities in other habitats.

This study tests the assumption that invaded habitats have limited ecological value by quantifying bird communities occupying *Acacia* thickets across a transformed landscape in the Western Cape Province of South Africa, specifically in the Cape Winelands and Cape Town Municipal areas. To do this I estimated the species richness, abundance, biomass and functional characteristics for communities supported by this novel habitat. Furthermore, I examined how patch and landscape level characteristics of *Acacia* thickets influence bird occupancy of patches within a highly transformed landscape. By quantifying the bird species richness, the density and the biomass within *Acacia* thickets at broad spatial scales, I provide base-line community data for bird assemblages within *Acacia* thickets. This will allow for better evaluation of the relative suitability of *Acacia* habitat for native birds compared with native vegetation and other transformed habitat types.

Methods

Study area

The study was conducted in the south-western region of the Western Cape Province, South Africa. The region falls within the Cape Floral Region (CFR), the smallest of the floral kingdoms and, due to high species richness and endemism, a biodiversity hotspot (Rebelo *et al.* 2006). The majority of the CFR is comprised of the Fynbos Biome and the indigenous lowland vegetation is characterised by vegetation types such as Renosterveld, Cape Flats Sand Fynbos and Coastal Standveld. Lowland vegetation is highly transformed by agriculture, urbanization and woody invasive species (Rouget *et al.* , 2003) and the remaining, endangered vegetation is largely restricted to protected areas (Rebelo *et al.* 2006).

Invasive Australian *Acacia* form one of the biggest threats to native biodiversity in the Fynbos Biome, with *Acacia saligna* and *A. cyclops* as two of the most widespread and problematic species (Rebello *et al.* 2006). In the Fynbos Biome, *Acacia* invasions cover 36% of the lowland areas, where they alter the physical structure of habitats by forming dense woody thickets and alter the floristic composition by out competing native vegetation (Holmes and Cowling, 1997). Underneath *Acacia* canopies, abiotic conditions are altered as soil nutrient levels change, leaf litter increases, surface soil moisture increases and the return rate and intensity of fire increases (Musil and Midgley, 1990). The accumulation of large seed banks, which germinate *en masse* after fire, and rapid seedling growth enhanced by nitrogen fixation, create positive feedback loops which maintain *Acacia* dominated systems (Le Maitre *et al.* 2011). *Acacia* were initially introduced for soil stabilization and fuel wood in the 19th century. Today, they continue to be used for these purposes but also provide an important source of income for previously disadvantaged peoples living in informal settlements in the region (Kull *et al.* 2011). In the Western Cape, 2,773,499 ha have some *Acacia* presence with most records coming from the Fynbos Biome (Kotzé *et al.* 2010). *Acacia*-dominated habitats can form very dense thicket, to more open savanna structure with an invasive grass understory, neither of which characterise indigenous lowland fynbos/renosterveld vegetation. The lack of a native analogue habitat in fynbos lowlands and the significantly altered ecological functioning of *Acacia* thickets creates a novel ecosystem for native animal species (Hobbs *et al.* 2006).

Bird survey

Bird surveys were conducted using the point count method, which is the most appropriate for dense vegetation, where walking transects is not possible (Bibby *et al.* 2000). All birds seen or heard within the *Acacia* thickets were recorded for estimates of species richness. For estimates of density I recorded all birds seen or heard within 60 m of point count locations. I identified individuals to species based on visual or auditory identification. The distances from the point to visually sighted birds were measured with a laser range finder and birds that were heard, but not seen, were judged to be within one of seven distance intervals from the point. Distance intervals were 0-7 m, 7-15 m, 15-25 m, 25-35m, 35-45 m and 45 – 55m and 55-65m. Distance intervals further from the point were larger to compensate for diminishing detection ability and distance estimation accuracy, which occurs as birds are recorded further from the point (Buckland *et al.*, 2004). Distance measurements for all birds seen or heard were used to generate estimates of bird density.. Point count surveys were conducted by one observer to minimize observer bias. Points were surveyed for six minutes, with a three-minute resting

phase preceding the count to allow birds to settle and resume normal behaviour (Bibby *et al.* 2000). Surveys were conducted over three-and-half hours starting at sunrise and the order in which points were visited within a site was rotated to minimize daily variation in bird activity. Surveys were not conducted on windy, misty or rainy days. Points were located 100 m from the habitat edge to minimize the influence of edge effects on bird activity and at least 200 m apart to insure independent samples of bird density (Buckland *et al.* 2004). The point count location was chosen by randomly placing a grid of points over a map of the study region and randomly choosing ten points that fell within each site. All sites had evidence of human disturbance, such as woodcutting, poaching and building of temporary human shelters. Points were not selected if they fell within 200 m of human shelters and during the course of the survey, points were no longer used if evidence of wood cutting was found within 100 m of the point. Sites were visited six times in a season and the order in which sites were visited was rotated to minimize weekly variation in bird activity due to weather. Seasonal surveys were conducted during February to March (autumn), June to July (winter), and September to December (spring), in 2010, to capture variation in bird communities with season. Additional sites were added in spring as this coincides with the peak breeding for many species (Hockey *et al.* 2005). For analysis, I treated data from different seasons and individual sites as independent samples of species richness and density – hereafter “survey” refers to seasonal site surveys.

Vegetation Surveys

Surveys were conducted using a modified version of James and Shugart (1970) plots around each point to capture variation in habitat structure. Vegetation was measured around each point in three, 30 m transects radiating out from each point. Stem density was measured by counting all stems one and half m in height, that fell within 1 m, on either side of the transect line. Vegetation height profiles were recorded by placing a graduated pole, with 10 cm demarcations, vertically every 2 m along the transect and recording the height at which the vegetation touched the pole. Ground cover type was recorded every 2 m where the pole hit the ground and categorized as bare dirt, rock, leaf litter, grass or herb. Canopy cover was recorded by counting presence or absence of canopy every 2 m along the transect using a canopy spotting scope (made of cardboard tube with crosshairs fitted at one end; James and Shugart 1970). Vegetation surveys took place over the summer months, November to January, 2010.

Site selection

Potential survey sites were identified using satellite images from Google Earth and with the help of local knowledge. All sites were mapped and verified to be *Acacia*-dominated by visiting them in person. A range of sites for the bird study was selected *a priori* to fall within different landscape contexts (Figure 1). Final site selection was based on total patch area (sites selected were greater than 36 ha to allow space for at least 10 point counts), landowner permission for access and site safety. Sites which experienced fire or significant wood harvesting were not used in subsequent surveys. Site characteristics can be found in Table 1.

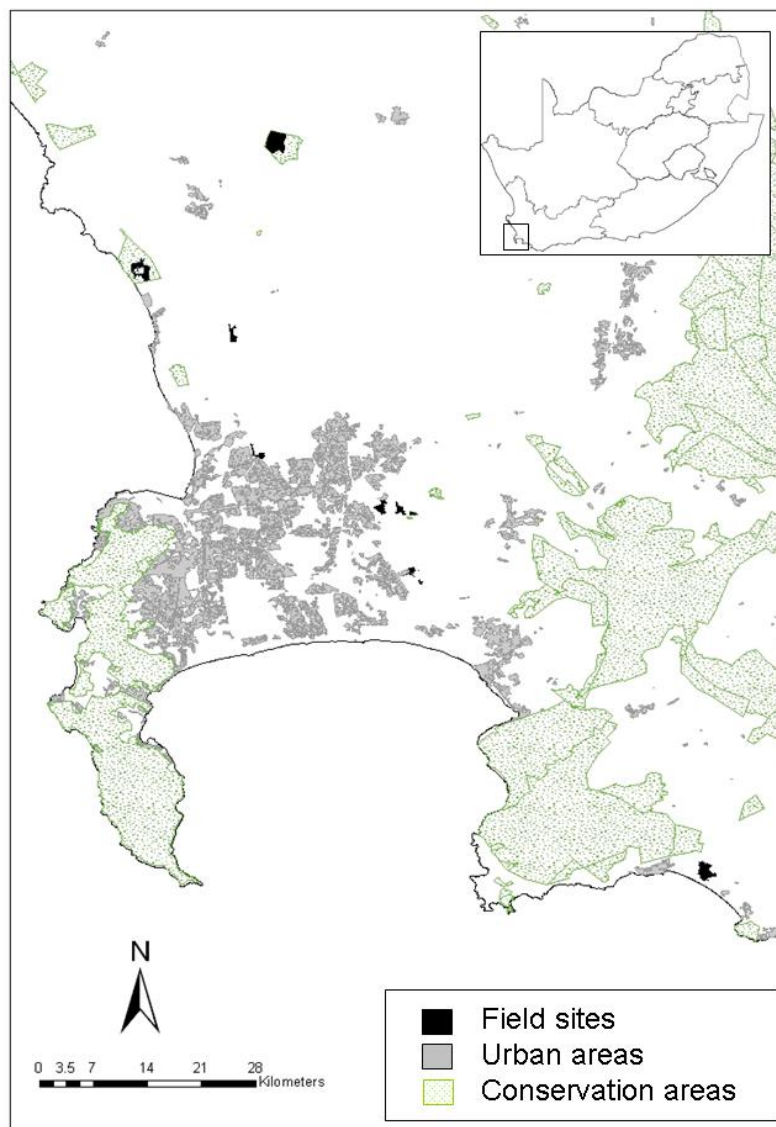


Figure 1. Map of the study region showing the approximate cover of urban and protected areas. Most protected areas are in the mountain regions while most urban development is in the lowlands. Field sites (black) spanned a range of landscape contexts to capture variation in bird occupancy of a site due to surrounding land use type. A description of study sites can be found in Table 1.

Table 1. Site descriptions and vegetation characteristics. Woody invasive species includes all invasives recorded on site, but in all cases *Acacia* species were the dominant invaders.

Site	Name	Manager	Area (ha ²)	Woody invasive species	Native vegetation	<i>Acacia</i> cover
1	Koeberg NR*	Eskom	306.82	<i>Acacia cyclops</i> , <i>A. saligna</i>	Coastal Strandveld	medium
2	Rooisands NR*	Cape Nature	302.88	<i>Acacia cyclops</i> , <i>A. saligna</i> , <i>Pinus spp</i> , <i>Leptospermum laevigatum</i>	Coastal Strandveld, Renosterveld	medium
3	Penhill Estates	Private	160.55	<i>Acacia saligna</i> , <i>Eucalyptus spp.</i>	Cape Flats Sand Fynbos	high
4	Zevenwacht Estates	Private	136.00	<i>Acacia saligna</i> , <i>Eucalyptus spp.</i>	Cape Flats Sand Fynbos	high
5	Kerk te Koe Farm	Private	122.54	<i>Acacia cyclops</i> , <i>A. saligna</i> , <i>Leptospermum laevigatum</i>	Cape Flats Sand Fynbos	high
6	DeGrendel Estates	Private	90.04	<i>Acacia saligna</i> , <i>Eucalyptus spp.</i>	Cape Flats Sand Fynbos	high
7	Riverlands NR*	Cape Nature	499.52	<i>Acacia saligna</i>	Cape Flats Sand Fynbos	low
8	Bottelary Conservancy	Private	119.35	<i>Acacia saligna</i> , <i>Pinus spp.</i>	Renosterveld	low

*NR= Nature Reserve

Analysis

Species richness was calculated using Jackknife 2 (SJack2) estimates in EstimateS (Colwell, 2009). The second order Jackknife estimator has advantages over other estimators because it makes no assumptions about underlying species distributions and has been shown to be the most precise given small sample sizes (Hellmann and Fowler, 1999; Magurran, 2004). Richness and its variance were computed by bootstrap, sampling with replacement using 500 randomizations (Colwell, 2009). Total richness was calculated using data pooled from all sites, and site richness was calculated based on seasonal surveys. Differences in survey richness were assessed using Kruskal-Wallis rank sum test in R (R Development Core Team, 2011).

Total bird density was calculated using point count data and the program Distance (Thomas *et al.* 2010). Distance data from each point count was pooled over the six visits for each site during each season and analysis used the Covariate Distance Sampling engine to determine differences in detection probability of birds for each survey. Detection functions were fitted to the data and final model selection was determined using Akaike's information criterion (AIC) (Buckland *et al.* 2004). Final estimates and their variance were generated using a uniform model with simple polynomial adjustment term in which the samples were bootstrapped 999 times (Buckland *et al.* 2004). The fit of the model indicated that detection probability was low beyond 45 m, therefore distance data used for the analysis of density, biomass, and functional group abundance was truncated at this distance. This procedure estimated density and total number of individuals per survey as well as average density and total individuals across all sites. Differences in bird density between surveys were assessed using Kruskal-Wallis rank sum test in R (R Development Core Team, 2011).

Feeding guild density was estimated by post-stratification of the pooled data from all surveys; the low abundance of some feeding guilds required pooling to obtain the number of observations (60–90) needed to produce robust density estimates. Species were grouped into six feeding guilds – frugivore, granivore, insectivore, mixed, nectarivore, and predator – using diet descriptions in Hockey *et al.* (2005). Differences between groups were assessed using Kruskal-Wallis rank sum test in R (R Development Core Team, 2011).

The avian biomass supported by each site was also estimated as the total biomass observed within a known area around each point count per visit. Total biomass around each point was the total mass of individuals observed within 45 m of point counts, calculated as the density for each species by the mean mass of each species summed for the assemblage at each point. The mean of the all points per survey was then used to estimate mean biomass per patch. Individuals were assigned species-appropriate mean body mass from Hockey *et al.*

(2005). Species body masses were calculated using methods described below in the analysis of BSFD. The biomass of each survey was compared using Kruskal-Wallis rank sum test. Total survey biomass was calculated by multiplying the estimate of biomass by total site area.

Species abundance within and across sites was evaluated using rank abundance curves and Simpson's Index. Rank abundance curves show the proportional contribution each species makes to the total number of individuals observed. Abundance values for each species use observations of abundance from point counts. Evenness was measured for each survey assemblage using the inverse Simpson's index in the package BiodiversityR (Kindt and Coe, 2005).

Further, to identify species that were indicators of *Acacia* thickets, indicator values were calculated for each survey (Dufréne and Legendre, 1997). Indicator values reflect the relative abundance and the relative frequency of a species within a site, with the highest value obtained when a species is abundant, widespread and unique to a site (McGeoch and Chown, 1998). Species indicator values and significance were calculated using 999 iterations of survey samples, using survey visits as samples. Species with an indicator value greater than 75% are considered to be representative species for each survey. Indicator values were calculated for the entire study to identify species representative of *Acacia* thickets. Additionally, indicator values were calculated for three levels of *Acacia* cover by grouping surveys by the level of *Acacia* cover at each site (Table 1).

Vegetation structure and habitat transformation can affect the body size frequency distribution (BSFD) of birds and other organisms occupying a site (Polo and Carrascal, 1999; Brown, 2007; Chown and Gaston, 2010). To test if altered habitat structure favoured certain body sizes I examined the BSFD for the entire *Acacia* assemblage and compared it to the regional BSFD. The regional BSFD of terrestrial species was generated using a species list for the Western Cape from the South African Bird Atlas Project (Harrison *et al.* 1997), with seabirds and aquatic birds removed. The BSFD for *Acacia* was generated using the total species observed across all sites. Body mass was used as a surrogate for body size and mass measurements were obtained from Hockey *et al.* (2005). Body size of individual species was calculated using the mean body mass reported for adult birds. If mass differed between males and females the mean was calculated and used. If body masses were reported as a range, the mid-point of the body mass range was used. If the mean body mass for either sex was taken from less than ten individuals and a mean mass was available from a large number of unsexed individuals, the unsexed mean was used (see Greve *et al.* 2008). Body mass was \log_{10} transformed then frequency distributions were generated by plotting the number of species occurring in equal sized body-classes (Bakker and Kelt, 2000; Greve *et al.* 2008).

To further test if birds with certain body sizes favoured *Acacia*, the median body size was calculated for each survey assemblage. Median body mass was used instead of mean body mass due to the highly right-skewed frequency distribution of body sizes across individuals (Appendix I). Individuals observed within 45 m of point counts were assigned species-appropriate mean body mass from Hockey *et al.* (2005). Species body masses were calculated using methods described above in the analysis of BSFD. The median mass was then compared using Kruskal-Wallis rank sum test.

Total vegetation cover and vegetation productivity was measured for each site using satellite images of enhanced vegetation index (EVI) scores. EVI, similar to normalized difference vegetation index (NDVI), measures light reflectance from chlorophyll to evaluate total leaf area, but EVI measures additional light colour bands making it more sensitive to canopy density and structural variation in vegetation and allows for better evaluation of vegetation densities (Huete *et al.* 2002). Data was used from NASA's Terra satellite, which produces EVI images with 250 m by 250 m pixel resolution every 16 days. EVI images were projected over maps of the study sites and the value of each EVI pixel that fell within the site was counted as a sample of vegetation density. EVI was compared across months and across surveys using a Kruskal-Wallis rank sum test.

The influence of site characteristics on bird communities was tested with generalized linear models (GLM) in R (R Development Core Team, 2011). Explanatory variables included stem density, canopy cover, mean vegetation height, average EVI value, distance to urban areas, distance to indigenous vegetation and total habitat area. Response variables tested were species richness, density, body size and biomass. Explanatory variables were included in final models if they had a variance inflation factor of less than four to account for collinearity. The final model for species richness included the explanatory variables stem density, mean vegetation height, average EVI value and distance to urban areas. The final models for density and biomass included stem density, canopy cover, mean vegetation height, average EVI value, distance to urban areas and distance to indigenous vegetation. Following recommendations from Zuur *et al.* (2009), I used both Poisson (scaled to compensate for overdispersion) and negative binomial error distributions and model simplification was performed using the "drop1" function (for Poisson errors) or "step" function (for negative binomial errors) in the MASS package (Venables and Ripley, 2002).

To estimate the total number of birds and the total biomass of birds supported by *Acacia* thickets at landscape scales, bird density and biomass estimates from this study were multiplied by estimates of total *Acacia* cover for the CFR found in the literature. The total number of individuals was calculated using a range of density estimates from each survey.

Bird biomass present in the invaded area in the CFR was calculated using estimates of biomass from surveys. Individual density and biomass estimates from this study were multiplied by estimates of the total area invaded by *Acacia* in the CFR lowlands (Rebelo *et al.* 2006) and for the entire Fynbos Biome (Kotzé *et al.* 2010).

Results

Surveys took place over 96 days in the field. A total of 336 hours were spent surveying in *Acacia* thickets. A total of 5,954 individual birds were recorded. In total, 75 species were observed across all surveys (data on species abundances, feeding guilds, and scientific names are shown in Appendix I). Sample-based rarefaction curves estimating species richness for the entire study reached asymptotes, suggesting that the survey captured most species utilizing *Acacia* thickets (Figure 2). Different estimates did not converge but levelled off with a difference of 12 species (Mao Tau = 68 species, SJack2 = 80 species).

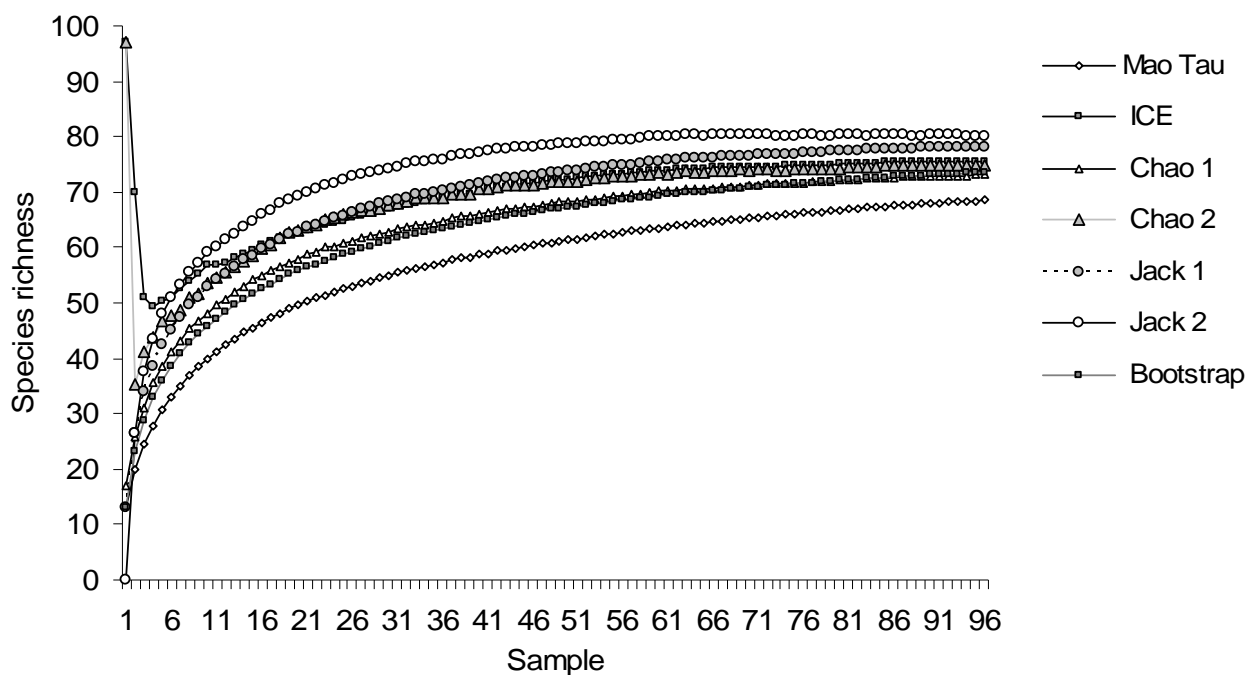


Figure 2. Species richness estimates for the entire study. All estimates show signs of levelling off, although Mao Tau, ICE, and Chao estimates do not converge.

Mau Tau species accumulation curves for each survey did not level off (Figure 3) suggesting interpretation of survey richness should be made with caution. However, accumulation curves for all sites showed similar shapes suggesting that survey effort was consistent across sites.

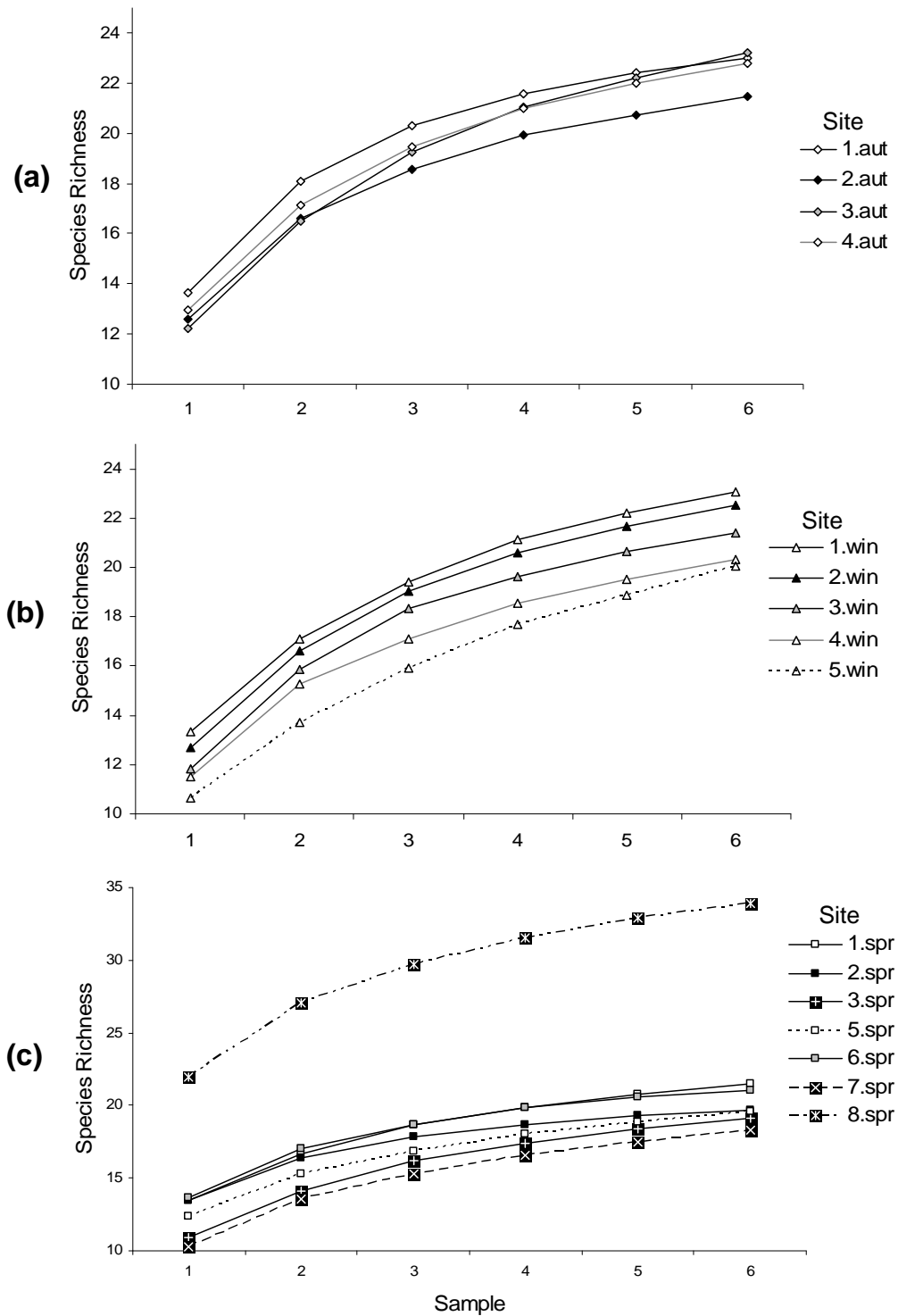


Figure 3. Mao Tau sample based rarefaction curves for each site in autumn (a), winter (b), and spring (c) did not reach asymptotes. All curves displayed similar shapes indicating that survey intensity was similar between sites.

Comparisons of species richness estimates, using the second order Jackknife (Figure 4a), calculated for each survey independently, found differences between sites within one season, between sites across seasons and for the same sites across seasons. Species richness for each survey was low relative to the entire study, but remained largely the same across seasons. Estimates ranged from 23 – 40.8 species. The mean survey richness was 29 species. Site 8, which was only surveyed in spring, was the least disturbed site and had the highest estimate of species richness (40.8 ± 0.341 : Table 2). The high variation in estimates for each survey (i.e. the spread of the whiskers on the box plot and the presence of outliers) is due to the presence of species that were observed only once or twice during each survey.

Density estimates for each survey varied significantly across sites and seasons (Figure 4b). Surveys conducted at the same site during all three seasons (sites one, two and three) had highest bird density in spring, intermediate in autumn and lowest in winter. The highest density of birds was recorded at site eight in spring (21.1 ± 0.0718 birds.ha⁻¹) and the lowest density was recorded in spring at site seven (3.12 ± 0.1339 birds.ha⁻¹). The estimated total number of birds supported during each survey ranged from 584 ± 0.898 birds, at site five in winter, to $3,008 \pm 21.9$ birds at site one in autumn (Table 2). A total of $26,355 \pm 37.3$ birds were estimated to have occupied the surveyed habitat patches over the course of the study (Table 2).

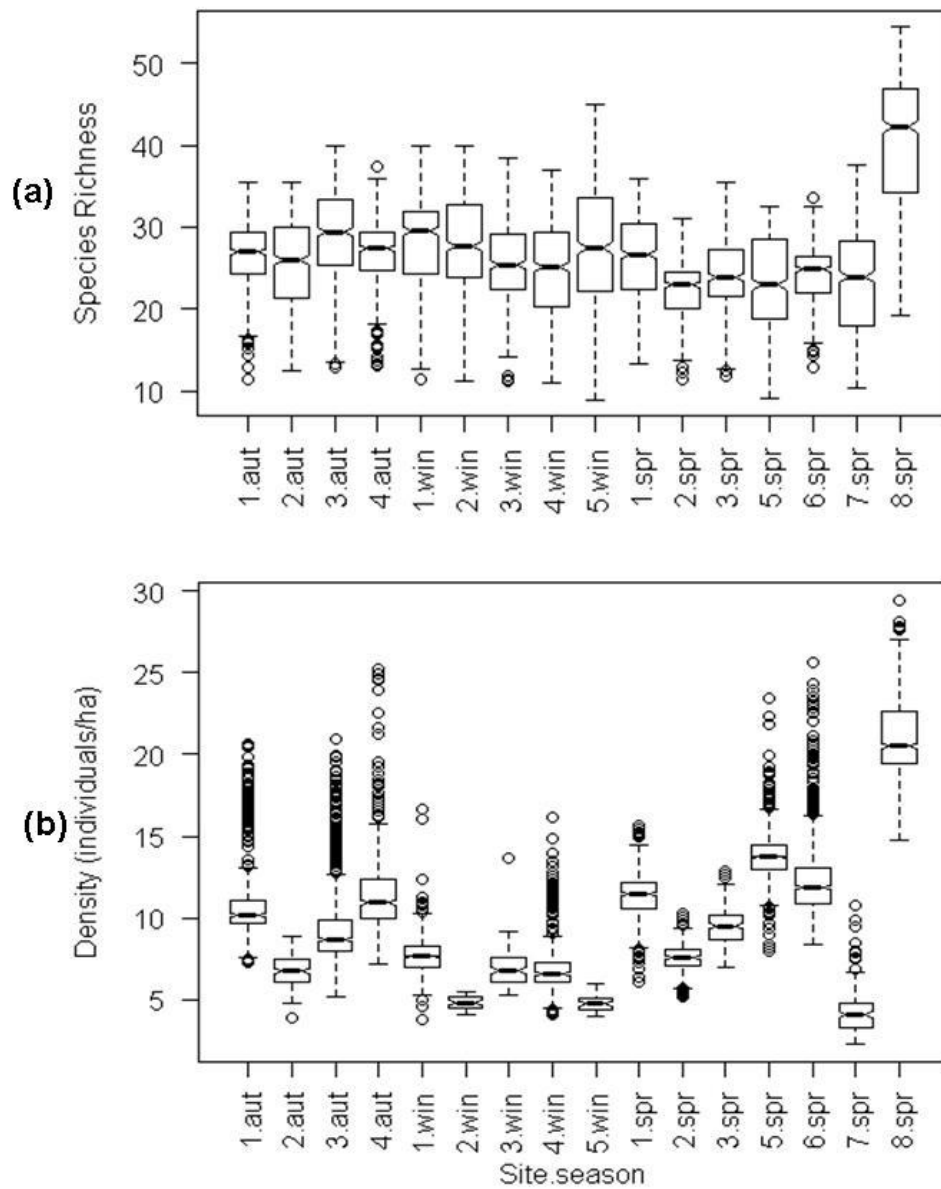


Figure 4. Bootstrapped SJack2 species richness estimates (a) showed significant differences between surveys (Kruskal-Wallis chi-squared = 1828.576, $df = 15$, $P = 2.2E-16$). Mean bird density (b) also showed significant differences across surveys (Kruskal-Wallis chi-squared = 7166.191, $df = 15$, $P < 2.2E-16$). Horizontal lines in the boxes represent median values, and lower and upper box boundaries indicate 25th and 75th percentiles. Whiskers span two standard deviations of the mean and points beyond the whiskers are outliers. Non-overlapping notches in boxes indicate significantly different medians.

Density estimates of feeding guilds showed significant differences in the abundance of feeding guilds (Kruskal-Wallis chi-squared = 6797.67, $df = 6$, $P = 2.2E-16$). Mixed feeders (3.98 ± 0.00912 birds.ha⁻¹) are by far the most abundant feeding guild in *Acacia* stands followed by insectivores (3.01 ± 0.010755 birds.ha⁻¹) and granivores (1.03 ± 0.00349 birds.ha⁻¹; Figure 5). The low numbers of frugivores (0.438 ± 0.00161 birds.ha⁻¹), nectarivores and predators makes fitting density detection functions more difficult and estimates of density for these guilds should be treated with caution.

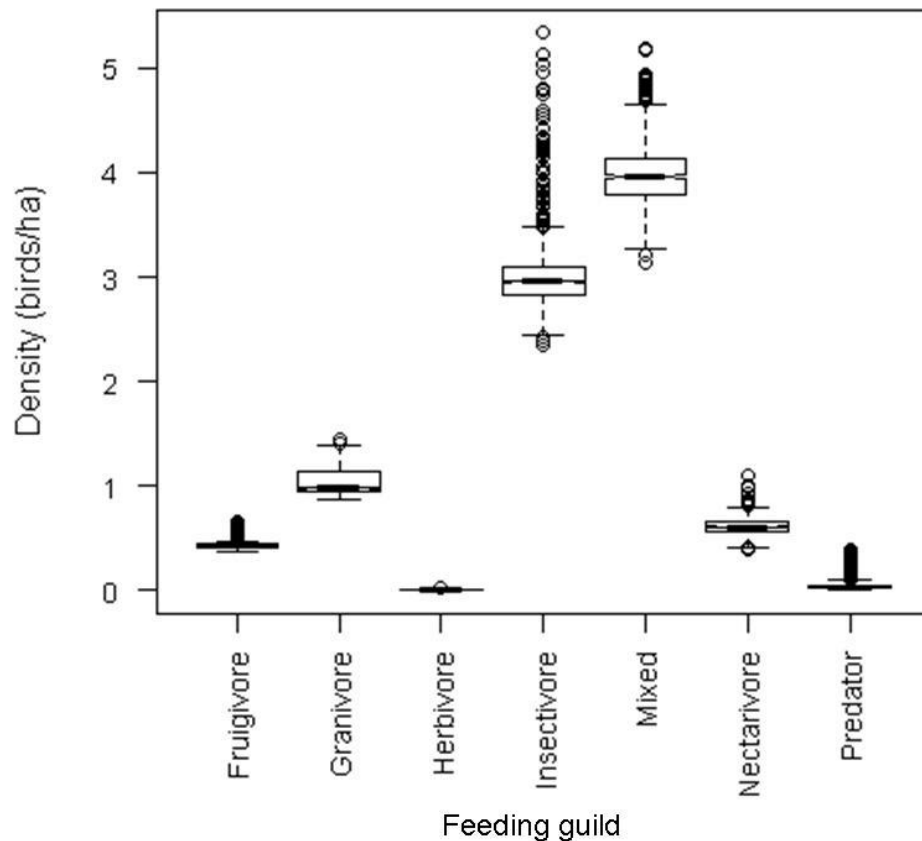


Figure 5. The density of birds in each of the feeding guilds for the entire study region showed significant differences across sites (Kruskal-Wallis chi-squared = 6797.67, $df = 6$, $P = 2.2E-16$). Densities were calculated based on guild abundance across all sites. Due to the low abundances of most feeding guilds estimates were generated using all observations from all sites. Horizontal lines in the boxes represent median values, and lower and upper box boundaries indicate 25th and 75th percentiles. Whiskers span two standard deviations of the mean and points beyond the whiskers are outliers. Non-overlapping notches indicate significant differences between medians.

Rank abundance curves for the entire study (Figure 6a) and for seasonal surveys (Figure 6b, c, d) showed that communities were highly dominated by five species. The most abundant species were (proportion of total observations) Karoo Prinia (19.1 %), Cape Robin Chat (15.2 %), Cape White-eye (14.3 %), Cape Canary (9.52 %) and Cape Bulbul (5.35 %).

Together these five species made up 63.5 % of all observations..The inverse Simpson’s index showed that that most sites had similar evenness (Table 2). Site eight was had the highest evenness in spring (11.71) while the lowest evenness was sampled at site four in winter (5.69; Table 2).

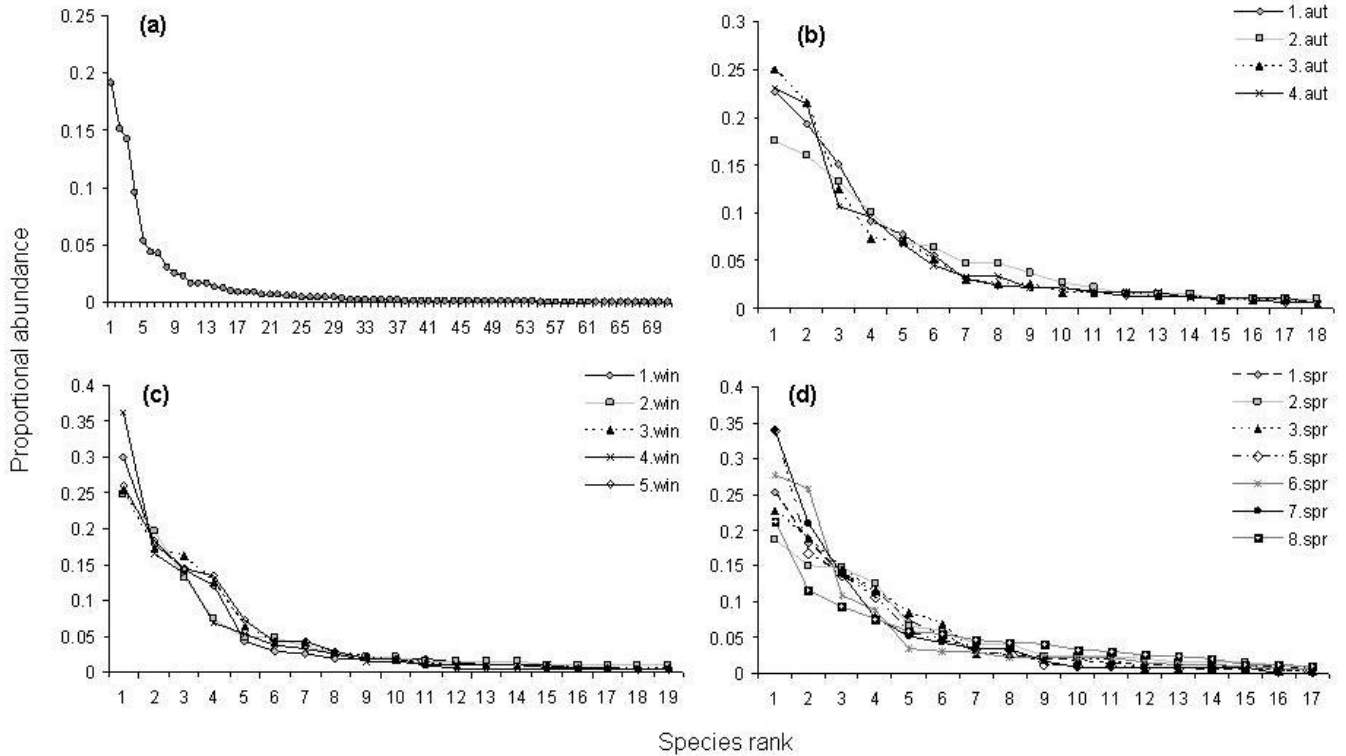


Figure 6. Rank abundance curves for total birds observed at all sites (a), for autumn surveys (b), winter surveys (c) and spring surveys (d). Most assemblages were highly dominated by three or four species. Across all sites (a), the five most abundant species (and percentage of total observations) were Karoo Prinia (19.1 %), Cape Robin Chat (15.2 %), Cape White-eye (14.3 %), Cape Canary (9.52 %) and Cape Bulbul (5.35 %). Legends contain survey codes with the site number and season abbreviation.

1 **Table 2.** Assemblage characteristics for each survey and all sites pooled. Estimates of density and number of individuals represent the mean
2 bootstrapped values from the program Distance (Thomas *et al.* 2010). Species richness estimates are the mean bootstrapped SJack2 values from
3 EstimateS (Colwell, 2009). Evenness was calculated using the “BiodiveristyR” (Kindt and Coe, 2005) package in R (R Development Core Team,
4 2011). Median body sizes were calculated using data from Hockey *et al.* (2005) and the South African Bird Atlas Project. Median body size and
5 biomass for each assemblage calculated by assigning species-appropriate mean body masses, from Hockey *et al.* (2005), to individuals observed within
6 45 m of point counts. Total biomass is calculated by multiplying the mean biomass for each survey by the total patch area for each site.

7

Survey	Site	Density (ha) ± SE	Total individuals ± SE	SJack2 ± SE	Evenness (1/D)	Median body mass (g)	Biomass ± SE (g/ha)	Total biomass (kg)
1	1.aut	9.8 ± 0.0873	3,008 ± 21.9	26.4 ± 0.178	7.6	28.4	396 ± 100	121
2	2.aut	5.98 ± 0.0643	1812 ± 5.58	25.4 ± 0.227	10.05	17.45	239 ± 60.1	72.5
3	3.aut	9.35 ± 0.0843	1501 ± 13.5	29.1 ± 0.258	7.26	26.8	166 ± 23.5	26.5
4	4.aut	11.3 ± 0.0693	1544 ± 9.41	26.8 ± 0.204	7.98	26.8	193 ± 38.6	26.2
5	1.win	7.27 ± 0.0423	2,232 ± 12	27.9 ± 0.264	6.4	28.4	147 ± 17.7	44.9
6	2.win	4.81 ± 0.0582	1,456 ± 2.24	27.9 ± 0.268	7.98	10.9	94 ± 11.6	28.2
7	3.win	6.24 ± 0.1115	444 ± 1.10	25.7 ± 0.226	7.25	15.2	321 ± 140	51.5
8	4.win	5.82 ± 0.0826	791 ± 7.00	24.9 ± 0.261	5.69	14	207 ± 57.9	28.1
9	5.win	4.77 ± 0.0619	584 ± 0.898	27.5 ± 0.368	7.14	15.2	135 ± 26.7	16.4
10	1.spr	11.31 ± 0.0409	3,471 ± 12.7	24.2 ± 0.221	7.46	20.6	388 ± 90.3	119
11	2.spr	6.71 ± 0.039	2,033 ± 9.83	23.9 ± 0.144	9.72	10.9	147 ± 31.7	44.3
12	3.spr	8.47 ± 0.056	603 ± 2.24	23.5 ± 0.217	7.88	17.45	330 ± 79.3	53.0
13	5.spr	13.8 ± 0.046	1,691 ± 5.61	26.2 ± 0.245	5.82	15.2	227 ± 23.8	27.8
14	6.spr	12.3 ± 0.0725	1,107 ± 6.52	22.3 ± 0.163	6.39	15.2	138 ± 19.3	12.4
15	7.spr	3.12 ± 0.1339	1,559 ± 11.6	23.4 ± 0.307	6.12	10.95	102 ± 48.9	50.9
16	8.spr	21.1 ± 0.0718	2,518 ± 8.55	40.8 ± 0.341	11.71	20	303 ± 39.1	36.1
	All sites*	7.87 ± 1.11	26,355 ± 37.3	80.1 ± 0.381	na	15.20	224 ± 15.4	791

8 * Estimates for all sites calculated using pooled data from all surveys

Indicator values calculated for the entire study and for three levels of *Acacia* cover identified very few species as indicators (Table 3). Across all sites (Table 3a) three species had *Indval* scores higher than 75% although these values were not tested for significance. Across the three levels of *Acacia* cover (Table 3b) only the lowest level had species which had significant values above 70 %. The Grey-backed Cisticola and Yellow bishop were determined to be an indicators of habitats with low *Acacia* cover.

Table 3. The five species with highest indicator values for all sites (a) and for three levels of *Acacia* cover (b). Species are considered indicators of a particular habitat group if they have significant *Indval* scores above 70 %. * $P < 0.05$

(a)

All sites	
<u>Species</u>	<u>Indval</u>
Karoo Prinia	100
Cape Robin-chat	93.8
Cape White-eye	84.4
Cape Turtle Dove	63.5
Pied Crow	57.3

(b)

<u>Acacia cover</u>					
Low		Medium		High	
<u>Species</u>	<u>Indval</u>	<u>Species</u>	<u>Indval</u>	<u>Species</u>	<u>Indval</u>
Grey-backed Cisticola	89*	Lesser-double	48.3*	Cape Canary	57.4*
Yellow Bishop	71.3*	Collared Sunbird		Cape Turtle Dove	33.2
Yellow-billed Kite	54*	Cape Bulbul	45*	Fiscal Flycatcher	30.1
Karoo Prinia	43.7*	Cape Robin-chat	40.7*	Pied Crow	27.2
Karoo Scrub-Robin	43.6*	Cape White-eye	37.9	Bully Canary	25*
		Cape Batis	32.5*		

The BSFD distribution for *Acacia* assemblages when compared to the regional assemblage, differed in their variance (F-test: $F = 6.072$, $df = 2886$, $P < 2.2E-16$), but showed no differences in range and mean (Welch two sample t-test: $t = 1.82$, $df = 2550$, $P = 0.0684$; Figure 7). The difference variance is expected as the *Acacia* BSFD contained fewer species (72) compared with the region (292). The similarity in the ranges suggests that *Acacia* support a community with a functionally similar range of body sizes. The BSFD for the communities found in *Acacia* display a bimodal distribution similar to that of the regional BSFD (meaning birds of all sizes can use the habitat).

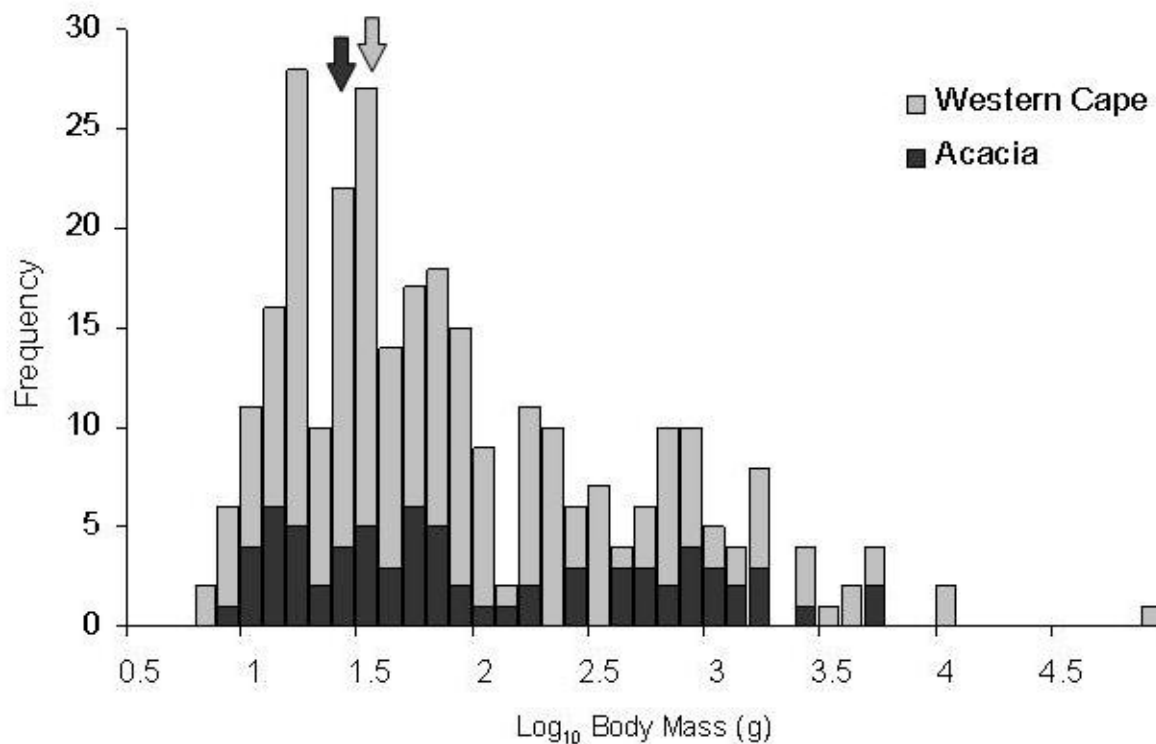


Figure 7. Species body size frequency distribution for log transformed body mass for the Western Cape (292 species, in grey) and for the species found in *Acacia* from this study (73 species, in black). Arrows mark mean Log body size for the Western Cape (1.56) and for *Acacia* (1.47) and are not significantly different between the distributions (Welch two sample t-test: $t = 1.82$, $df = 2550$, $P = 0.0684$). Variance was found to significantly differ between distributions (F-test: $F = 6.072$, $df = 2886$, $P < 2.2E-16$)

Assemblages had a range of median body sizes from 10.9 to 28.4 g and significant differences were found across surveys (Kruskal-Wallis chi-squared = 177.93, $df = 15$, $P = 2.2E-16$; Figure 8). Median body sizes observed during surveys showed significant differences between sites surveyed within a season, and between sites surveyed across seasons. Furthermore, most surveys had a median body size smaller than the mean body size for the entire *Acacia* assemblage, highlighting that *Acacia* thickets are dominated by small bodied birds.

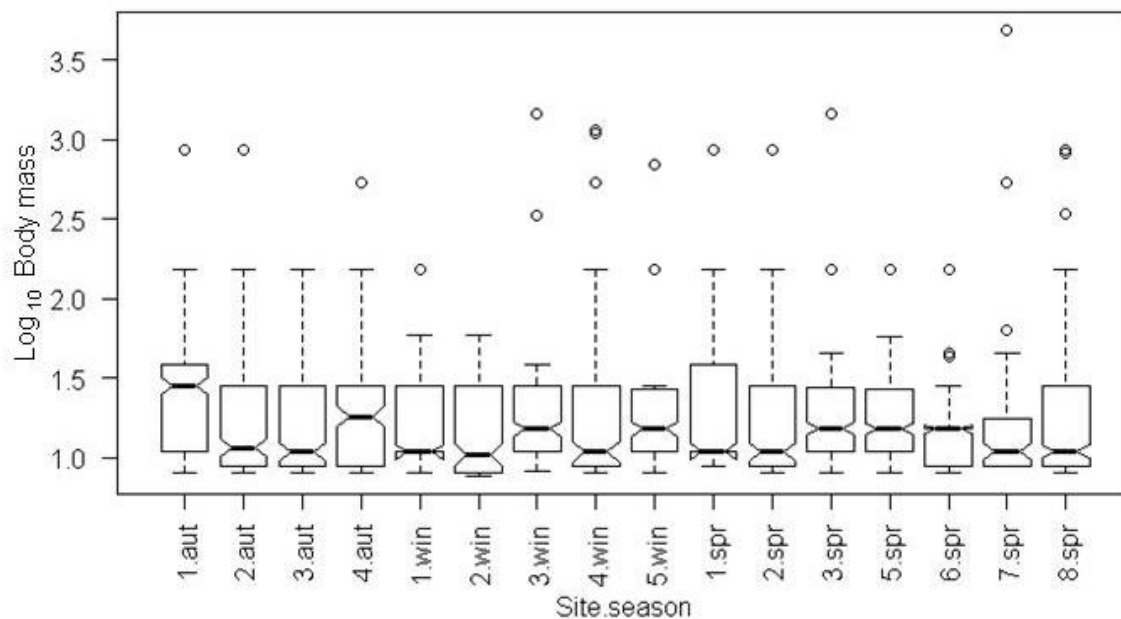


Figure 8. Log body mass for individuals observed in each survey. Significant differences are seen between the median body size of individuals across sites (Kruskal-Wallis chi-squared = 177.93, $df = 15$, $P = 2.2E-16$). Horizontal lines in the boxes represent median values, and lower and upper box boundaries indicate 25th and 75th percentiles. Whiskers span two standard deviations of the mean and points beyond the whiskers are outliers. Non-overlapping notches indicate significant differences between medians.

The biomass of birds varied across surveys within a season and between seasons (Kruskal-Wallis chi-squared = 150.3, $df = 15$, $P < 2.2E-16$; Figure 9). Mean biomass ranged from $94 \pm 11.6 \text{ g.ha}^{-1}$ to $396 \pm 100 \text{ g.ha}^{-1}$. The average biomass across all surveys was $224 \pm 15.4 \text{ g.ha}^{-1}$. Total estimated bird biomass for each study site varied significantly across surveys within one seasons and across seasons (Kruskal-Wallis chi-squared = 154.1155, $df = 15$, $P < 2.2E-16$; Table 2). Mean biomass of birds supported at each site were estimated to range from 12.4 kg to 121 kg. Across the entire survey period the study sites supported an estimated 791 kg of birds.

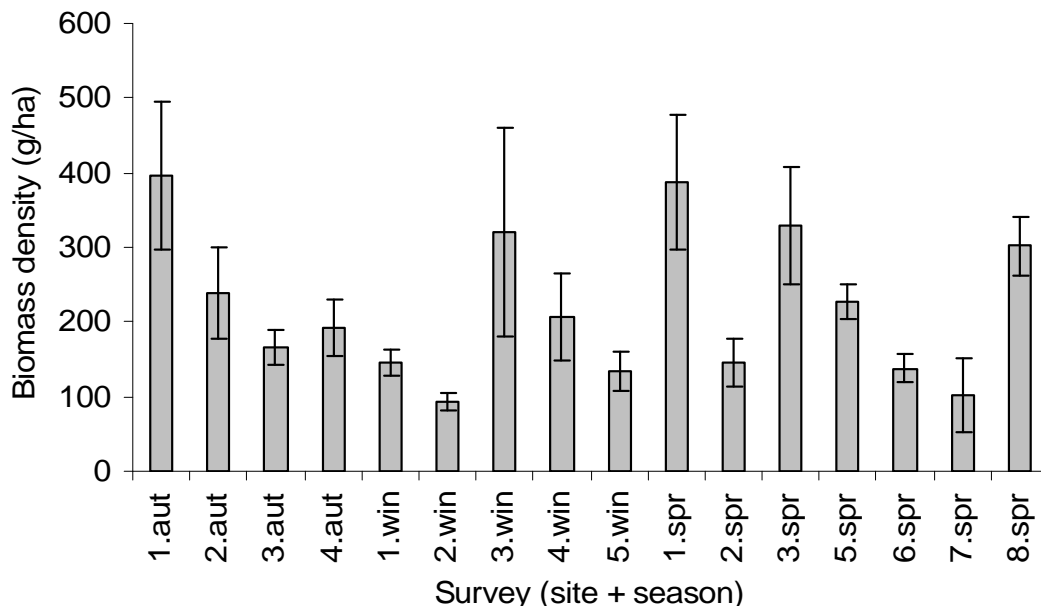


Figure 9. Average biomass across surveys varied significantly (Kruskal-Wallis chi-squared = 150.3, $df = 15$, $P < 2.2E-16$). There was no consistent pattern in variation with season, although sites that supported high avian biomass in one season tended to support high biomass across surveys. Biomass was calculated by assigning species-appropriate mean body mass, from the literature, to individuals observed within 45 m of point counts. Grey bars represent average values and error bars represent standard errors.

Vegetation characteristics for each site (Table 4) showed significant differences in mean vegetation height ($F_{6, 57} = 21.1$, $P = 7.013E-13$), stem density ($F_{6, 57} = 4.37$, $P = 0.00107$) and percentage canopy cover ($F_{6, 57} = 17.1$, $P = 3.67E-11$). Site six had the highest mean vegetation height (391 ± 3.85 cm) and site seven had the lowest mean vegetation height (70 ± 1.1 cm).

Table 4. Vegetation characteristics for each site. The *Acacia* stand surveyed at site 4 was cut down before vegetation surveys could be conducted. NR = Nature Reserve

Site	Name	Mean vegetation height (cm) \pm SE	Mean stem density (m ²) \pm SE	Mean canopy cover (%) \pm SE
1	Koeberg NR	110 \pm 1.85	1.23 \pm 0.295	13.1 \pm 3.43
2	Rooisands NR	200 \pm 2.42	2.32 \pm 0.488	32.6 \pm 4.34
3	Penhill Estates	308 \pm 4.73	0.869 \pm 0.145	32.7 \pm 3.74
4	Zevenwacht Estates	na	na	na
5	Kerk te Koe Farm	319 \pm 3.07	1.65 \pm 0.317	40.0 \pm 3.12
6	DeGrendel Estates	391 \pm 3.85	2.00 \pm 0.336	34.4 \pm 3.53
7	Riverlands NR	70 \pm 1.1	0.0179 \pm 0.00751	0
8	Bottelary Conservancy	153 \pm 1.96	0.555 \pm 0.129	5.36 \pm 2.67

Vegetation density and productivity as measured by EVI showed significant variation over the ten months of the study (Kruskal-Wallis chi-squared = 369.6, $df = 10$, $P < 2.2E-6$; Figure 10a) and between study sites (Kruskal-Wallis chi-squared = 1271.364, $df = 15$, $P < 2.2E-16$; Figure 10b). EVI values increased during the winter and spring months (June through October) which is the time of year which receives the most rain and is also the growing season for *Acacia*.

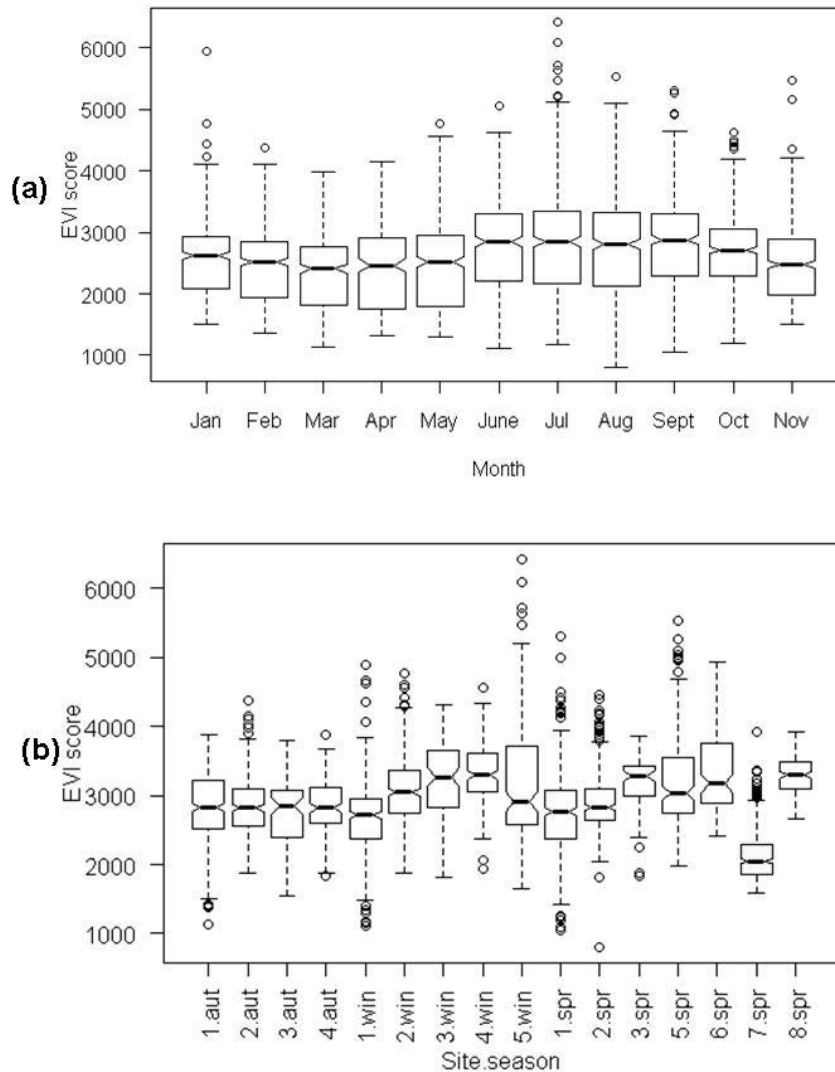


Figure 10. Monthly mean enhanced vegetation index (EVI) scores for all months (a) and for each survey (b). EVI values are measure total leaf area cover with higher EVI values indicating denser canopy cover and total vegetation cover. EVI values show significant differences between months (Kruskal-Wallis chi-squared = 369.6, $df = 10$, $P < 2.2E-6$) with highest values coinciding with the winter and spring months (June through October) which receive the most rain. Significant differences were found between surveys as well (Kruskal-Wallis chi-squared = 1271.364, $df = 15$, $P < 2.2E-16$), with greater variation between EVI scores for spring and winter survey than in autumn. Median scores are represented by the thick black line, non-overlapping notches indicate significant differences, box lines are 95 % confidence intervals and whiskers spread 2 standard deviations from the median.

The GLM with negative binomial errors gave the best fit minimal adequate models for all response variables (Table 5). Species richness showed a significant positive relationship with EVI (coefficient = 0.00039, t -value = 4.37, P = 0.0113) and a significant negative relationship with mean vegetation height (coefficient = -0.00134, t -value = -3.25, P = 0.00769). Density also showed a significant positive relationship with EVI (coefficient = 0.0014, t -value = 4.01, P = 0.0036), a negative relationship with canopy cover (coefficient = -2.02, t -value = -2.36, P = 0.0429) and a negative relationship with the winter season (coefficient = -0.628, t -value = -2.53, P = 0.0323). Median body size for each assemblage showed a negative relationship with canopy cover (coefficient = -2.76, t -value = -2.08, P = 0.037) and season (coefficient = -0.541, t -value = -3.359, P = 0.000781). Furthermore, median body size showed a non-significant positive relationship with mean vegetation height and this term was included in the model as it decreased total model variance. Biomass across surveys showed a positive relationship with EVI (coefficient = 0.00138, t -value = 2.75, P = 0.0252) and a negative relationship with stem density (coefficient = -0.475, t -value = -2.36, P = 0.0462). Biomass showed a non-significant positive relationship with distance to urbanized area and a non-significant negative relationship with spring and winter and these terms were included in the model as they decreased model variance.

Table 5. Results from models of the relationship between site level characteristics and species richness, density and biomass.

Model (error distribution)	Response variable	Deviance explained	Explanatory variables	coefficient	t -value	P
GLM (negative binomial)	Species richness	4.78/8.98 (53.2%)	Intercept	2.38	4.37	1.23E-05
			EVI	0.0003903	3.04	0.0113
			Mean vegetation height	-0.0013392	-3.25	0.00769
GLM (negative binomial)	Density	4.53/6.88 (65.9 %)	Intercept	3.34	4.98	6.23E-07
			EVI	0.0014	4.01	0.00306
			Canopy cover	-2.02	-2.36	0.0429
			Season (winter)	-0.628	-2.53	0.0323
GLM (negative binomial)	Median body size	38.02/51.58 (73.7 %)	Intercept	3.44	19.9	2.00E-16
			Canopy cover	-2.76	-2.084	0.0371
			Mean vegetation height	0.00251	1.489	0.136
			Season (spring)	-0.540765	-3.359	0.000781

Model (error distribution)	Response variable	Deviance explained	Explanatory variables	coefficient	<i>t</i> -value	<i>P</i>
GLM (negative binomial)	Biomass	18.6/32.7 (56.9 %)	(Intercept)	2.01	1.41	0.196
			EVI	0.00138	2.75	0.0252
			Stem density	-0.475	-2.36	0.0462
			Season (spring)	-0.46	-1.47	0.179
			Season (winter)	-0.755	-2.18	0.0606
			Distance to urbanized area	0.000185	1.75	0.118

Estimates of the total number of birds and total bird biomass supported by *Acacia saligna* and *A. cyclops* are listed in Table 6. The total area invaded is 2,773,498.75 ha for the entire Fynbos Biome (Kotzé *et al.* 2010) and 1,179,216 ha in the CFR lowland vegetation types (Rebello *et al.* 2006). Using the most conservative estimate of density from this study, *Acacia* stands support 8,653,316 birds across the Fynbos Biome, and 3,679,153 birds in the lowlands (Table 6). The highest estimates of individuals were 38,274,283 for Fynbos Biome and 16,273,181 for lowland *Acacia*. The low estimate of avian biomass within *Acacia* thickets are 257,935 kg and 109,667 kg for fynbos and lowlands respectively. High estimates of biomass are 1,101,079 kg for the fynbos and 468,148 kg for the lowlands.

Table 6. Estimates of the number of individuals (a) and biomass (b) supported by *Acacia* thickets. Low, average and high estimates of bird density were used to calculate the possible range for numbers of birds and biomass of birds for two estimates of *Acacia* cover. The estimates of *Acacia* cover are for the entire Fynbos Biome and just for the lowland regions within the Fynbos Biome.

(a) Total individuals

Estimate	Density estimates (birds/ha)	Total birds: CFR lowland <i>Acacia</i>	Total birds: Fynbos Biome <i>Acacia</i>
low	3.12	3,679,153	8,653,316
average	7.87	9,280,429	21,827,435
high	13.8	16,273,180	38,274,282

(b) Total biomass

Estimate	Biomass estimates (kg/ha)	Total biomass: CFR lowland <i>Acacia</i>	Total biomass: Fynbos Biome <i>Acacia</i>
low	0.093	109,667	257,935
average	0.224	264,144	621,263
high	0.397	468,148	1,101,079

Discussion

Species richness of the CFR bird communities generally decreases following habitat transformation (Armstrong and Van Hensbergen, 1994; Mangnall and Crowe, 2003; Fox and Hockey, 2007; Greve *et al.* 2011). However, the 75 species recorded in this study is close to the total number of species found by Dures and Cumming (2010) who found 79 species surveying birds in Lowland Sand Fynbos vegetation. They also found the Karoo Prinia and Cape White-eye to be the most common species across their sites, a result mirrored in the assemblages surveyed in *Acacia* from this study (Dures and Cumming, 2010). At the landscape scale, *Acacia* stands support a large number of native bird species, however, the high mobility of birds means that species could be recorded in *Acacia*, but not rely heavily on the habitat. High richness found across sites means that at the very least, these species are utilizing *Acacia* thickets to as temporary habitat space, making *Acacia* a potential corridor across landscapes. The arrangement and total area cover of vegetation at landscape scales directly affects how many species a region can support (Fahrig, 2003), and Radford *et al.* (2005) found that at a landscape scale, there is a minimum vegetation threshold required to support full suits of habitat dependant species. The total area of *Acacia* thickets and the high numbers of species recorded in them suggests that *Acacia* is contributing to total habitat area in a highly transformed landscape; however, the relative importance of *Acacia* compared with native vegetation is unknown.

Species richness for each survey was low (~ 25 species; Table 2) compared with total richness in this study. Except for site eight, most surveys found a similar number of species (between 23 and 29 species), and accumulation curves suggest that more species would be detected with longer surveys. The species accumulation curve for all sites had similar shapes suggesting that survey intensity was the same across sites allowing for comparison of richness estimates (Colwell, 2009). The differences between sites were expected as studies of fragmented habitats have shown that the interaction between landscapes and patch level characteristics creates high inter-patch variability in species richness (Bolger *et al.* 1997; Lee *et al.* 2002; Shanahan and Possingham, 2009). Excluding site eight, richness estimates across sites differed by less than six species, which is not a substantial difference considering the total richness for all sites. Species richness was only substantially higher at site eight (Figure 4a) which is managed as a reserve and contains large areas of mature native vegetation. Interestingly site seven had the lowest levels of *Acacia* invasion (Table 4; Appendix I) but did not support the highest number of species. This may be due to the fact that at site seven the natural vegetation was recovering from recent disturbance from both *Acacia* removal programs and previous land use practices (i.e. parts were ploughed for farming) that occurred

before the site was established as a reserve. The difference between site seven and eight highlights the importance of high quality habitat in supporting bird species and the relative value of *Acacia* thickets (as seen in the species richness of other sites) compared to degraded natural areas. The similarities between site seven and other more heavily invaded sites also suggests that for birds, degraded natural areas and *Acacia* thickets may be of similar habitat value.

Habitat structure can be an important driver of patch occupancy at landscape scales (Dean *et al.* 2002; Fleishman *et al.* 2009) and in this study species richness showed a negative relationship with mean vegetation height. This result is in line with the results of the study by Dures and Cumming (2010) who focused their surveys in native lowland vegetation the Cape. Mean vegetation height increases as *Acacia* become more dominant at a site. As *Acacia* become more dominant, they alter shade out native plant species reducing both biotic composition and structural diversity. *Acacia* thickets range in structure from very dense thicket to an open savanna, which seems diverse for large bodied terrestrial mammals (such as humans), however, the structure of a monoculture of *Acacia* trees may offer a structurally homogenized habitat for flying, small-bodied birds.

A positive relationship was found between species richness and EVI. Monthly variation in EVI reflects seasonality in vegetation productivity and total leaf area at a site. Canopy density is highest in the growing and flowering season (June – October), which coincides with winter rainfall in the Western Cape. Sites dominated by *A. saligna* (sites three, four and five) showed higher seasonal variation than sites dominated by *A. cyclops* (sites one and two). A weak positive relationship between species richness and EVI is likely the result of how resources abundance varies with vegetation productivity. Food resources that would vary with season include insect assemblages and annual plants. Foliage invertebrate assemblages on *Acacia* also show seasonal differences, notably in the abundance of insects (Procheş *et al.* 2008), although the importance of such variation in insect resources for birds requires further study (but see Fraser and Crowe, 1990). Additionally, seasonal variation in resources may be driven by annual grasses and herbaceous plant species emergence during the spring, possibly increasing floristic resource diversity and abundance within *Acacia* invaded areas.

The density of birds within a site can be a reflection of the resource abundance within a habitat (Bolger *et al.* 1997; Parrish *et al.* 2005). The high density of birds recorded at site eight was expected as it is the site with the lowest disturbance and with the highest richness (Table 2). The lowest density was found at site seven in spring, which was surprising as it was 600 m from the edge of the Tygerberg Nature reserve, a 300 ha reserve dominated by

renosterveld vegetation, although site seven was separated by vineyards and grazed pastures. The second and third lowest density of birds was observed in winter at sites two and five which was surprising as the sites had different surrounding land use and *Acacia* densities (Table 1,4). Bird density, similar to species richness showed a small positive relationship with EVI indicating that density increased with increasing seasonal resources in spring. Density also showed a negative relationship with canopy cover, which is likely a response to the closed canopy thickets formed by mature *Acacia* trees. This could be explained by the results of similar studies that found the highest densities of birds at intermediate levels of invasion of *Acacia* and *Tamarix* trees (Fraser and Crowe, 1990; van Riper III *et al.* 2008). At higher levels of invasion the habitats become structurally homogenized by invasive trees which form closed canopies. Density also showed a negative relationship with the winter season, which may be the result of fewer birds being present in *Acacia* thickets or the birds present may simply be less active, affecting detectability (Bibby *et al.* 2000; Symes *et al.* 2002). Most birds observed over all seasons were non-migratory species, although some birds do show seasonal variation in habitat preference at smaller spatial scales (Catry *et al.* 2003) and a more detailed study over multiple years would be needed to properly test seasonal differences in bird density.

The abundance of functional groups within *Acacia* habitat were dominated by two feeding guilds. The high abundance of mixed feeders and insectivores is likely the result of the high abundance of invertebrates that *Acacia* species can support (Procheş *et al.* 2008). The moderate abundance of granivores, relative to the other feeding guilds, may be explained by the presence of herbaceous annuals and invasive grasses which dominate the understory of many invaded areas, although the quality of invasive grasses as a food resource for native granivores has not been tested and warrants further study. However, some nectarivores were present at most sites suggesting that even this functional group can make use of *Acacia*. The only obligate nectarivore in the CFR is the Cape Sugarbird, which was not observed at any site. The other nectarivore species are not obligate nectarivores and are likely opportunistically taking advantage of both structural and invertebrate-food resources found in *Acacia* thickets. The low abundance of frugivores is interesting as *Acacia cyclops* is bird dispersed and forms a bright red, fleshy aril attached to its seeds. These arils are attractive to native species (Fraser, 1990; Underhill and Hofmeyr, 2007). However, *A. cyclops* does not flower or set seed en masse, and therefore may not provide enough rewards, consistently enough to support high densities of frugivores. The low density of predators, most of which were raptors, is not surprising given that point counts are not designed to efficiently sample large predators.

The evenness of communities reflects the diversity of resources within a habitat, with high resource diversity supporting more even assemblages. Habitat transformation can create habitats with few but highly abundant resources. Species that use remaining resources in altered habitat can become dominant species, lowering community evenness. Rank abundance curves for all observations as well as individual surveys show high proportional abundance of four to five species (Figure 6). The dominant species, Karoo Prinia, Cape Robin Chat, Cape White-eye, Cape Canary and Cape Bulbul are habitat generalists, which are also common garden species (Hockey *et al.* 2005). These species belong to the most abundant feeding guilds, the mixed feeders (Cape Robin-chat and Cape White-eye), insectivores (Karoo Prinia) and granivores (Cape Canary). The Cape Bulbul is a frugivore, whose high proportion in all observations is unexpected and was the most abundant frugivore across all sites. The inverse Simpson's index value measured evenness and its value increases as community diversity increases. Sites two, in autumn, and eight, in spring, were the most even sites, which was surprising because there are large differences in the richness and abundance of bird species found in each site (Figure 4: Table 2). The patterns generating evenness in the two sites were different as site two had a low estimate of richness (25.4 ± 0.227) and density (5.98 ± 0.0643), while eight had a high estimate of species richness (40.8 ± 0.341) and high density (21.1 ± 0.0718).

The indicator values generated for all study sites identified Karoo Prinia, Cape Robin-chat and Cape White-eye as having *Indval* scores above 75 %, suggesting these species are both abundant and widespread across study sites. This is not surprising given the relative dominance of these species in the observed assemblages. Their high indicator values reflect the low evenness shown by the rank abundance curves. Areas with low *Acacia* cover had one significant indicator species, the Grey-backed Cisticola, while areas with medium and high levels of *Acacia* cover had no species with values high enough to be considered indicators of the habitat. The lack of indicator species for areas with high cover of *Acacia* is not entirely surprising as *Acacia* thickets are dominated by habitat generalist species, which are abundant in both invaded and uninvaded habitats. Grey-backed Cisticolas were highly abundant only at sites with low *Acacia* cover and, despite being insectivores, likely have other habitat requirements that make *Acacia* thickets unfavorable habitat. Fraser and Crowe (1990) found that, along a gradient of *Acacia* invasion, Grey-backed Cisticola abundance decreased with increasing invasion. They attributed the loss to reduced levels of short vegetation beneath *Acacia* canopies, which is the preferred foraging substrate for the that species. The contrasting response to invasion from species such as Grey-backed Cisticola (low abundance in invaded areas) and Karroo Prinia (high abundance in invaded areas), which both belong to the

insectivore feeding guild, highlights the complex species-specific responses to invasion. These results suggest the need for caution when making assumptions of habitat suitability for species, even those belonging to similar functional groups. Further species-specific studies are required to understand why certain species are favored or disadvantaged by habitat transformation.

The body size frequency distribution of all birds found in *Acacia* spanned a similar range as the BSFD for the Western Cape (Figure 7). This suggests that *Acacia* support a large functional range of the bird body sizes at landscape scales. Compared with the regional BSFD, the *Acacia* BSFD had proportionally fewer birds, especially in the medium body size classes, reflecting the reduction in species richness. Additionally, it is likely that if more sites across the entire CFR were surveyed, more bird species would be found, and the *Acacia* BSFD and the regional BSFD would be more similar.

The median body size of individuals observed showed significant differences between surveys with site one having consistently higher median body mass observed across three seasons (Figure 8). All sites were dominated by small bodied individuals. The lowest median assemblage body size was 10.9 g which is the body mass of the Cape White-eye, a small, foliage gleaning, mixed feeder. The highest median body size across surveys, 28.4 g, is the body mass of the Cape Robin-chat, which is primarily a ground-foraging mixed feeder. Most species recorded in *Acacia* thickets are resident, non-migratory species, so the lack of changes in the body sizes between seasons is expected. The influence of vegetation characteristics on median body mass of individuals (Table 5) showed a negative relationship between body mass and canopy cover. Low floristic diversity beneath *Acacia* canopies means that the most abundant food resource in sites with high *Acacia* cover are likely to be insects and seeds from small, annual plants. These resources likely favour bird species that can forage for insects in dense vegetation and can consume small seeds, and explains the dominance of small foliage gleaning mixed feeders. Median body mass also showed a negative relationship with spring. Seasonal shifts in resources abundance could change dominant species within assemblages in *Acacia* thickets across seasons, but longer-term studies are required to test if the relationship found here is consistent across years.

Biomass was highly variable, suggesting both seasonal and patch level differences in the avian biomass supported across sites. However, some sites consistently supported a higher biomass across seasons (Figure 9). Site one supported the highest biomass and this likely the result of the surrounding native vegetation, Coastal Strandveld, which is known for supporting a high abundance of birds (Fox and Hockey, 2007). The biomass observed across surveys showed a positive relationship with EVI and a negative relationship with stem

density. Seasonal shifts in vegetation productivity, as measured by EVI, could reflect fluctuations in resource abundance, which in turn may influence the total bird biomass supported by *Acacia*. Denser *Acacia* canopies, or higher productivity in the rainy months could provide more insect and floristics resources across sites, although this requires further study. The negative relationship between biomass and stem density suggests that as *Acacia* become more dominant at a site there are less resources available and that sites with higher *Acacia* cover supported lower amounts of birds. EVI was found to have significant relationships with species richness, density and biomass, thus studies on how resources in altered habitats vary with changes in EVI have the potential to contribute to remote sensing of habitat use by birds.

The total abundance and biomass for the total *Acacia* invaded area in the CFR are reasonable estimates of bird density in *Acacia* invaded areas, as sites were representative of the range of habitats invaded by *Acacia*. The estimates of density and biomass found in individual surveys in this study (density: 3.12 – 13.8 birds/ha; biomass: .093 - .397 kg/ha) fall within ranges reported elsewhere for Western Cape bird communities. In Coastal Strandveld vegetation Fox and Hockey (2007) found much higher bird densities ranging from 80.2 – 95.9 birds per ha. In comparison to the results of this study, *Acacia* thickets support fewer individuals, which may be expected as Coastal Strandveld vegetation supports some of the highest number of species and species abundances in the CFR lowlands (Winterbottom, 1970). In a study of *Acacia* invaded habitats on the Cape Peninsula Fraser and Crowe (1990) found bird densities of 3.3 – 5.0 birds per ha and biomass ranging from 37.6 – 709.9 g per ha. In comparison, the results presented here, the estimates of density overlapped and they found a greater range of bird biomass. Furthermore, Fraser and Crowe (1990) surveyed in a range of invaded habitats and found that *Acacia* supported twice the density of birds compared to Restionaceous Tussock Marsh, but equivalent densities were found in *Acacia* thickets and Mountain Fynbos. Greve (2006) found bird densities across pine plantations and Mountain Fynbos ranging from 28 – 42.5 birds per ha, which is larger than the highest estimates found in this study. While estimates of total number of individuals presented here is not biologically significant without further information on population dynamics, it does serve as a reflection of the contribution that *Acacia* is making in supporting indigenous species at the landscape scale. The biomass of birds presented here can serve as a benchmark for future work comparing native and invaded communities; however further comparisons are not possible here, as most habitats in the CFR do not have accurate estimates of community density. There is a need for better quantification of community abundance in both natural and transformed habitats if the relative value of *Acacia* thickets is to be fully understood.

Conclusion

The results of this study quantify the species richness, total abundance, feeding guild abundance, community evenness, body size distributions and biomass of bird communities occupying *Acacia* thickets in the lowlands of the CFR. These results describe avian assemblages over the course of three seasons, presenting a snapshot of community dynamics which are likely to fluctuate on longer temporal scales (Hurlbert and Haskell, 2003). *Acacia* thickets support relatively diverse bird assemblages considering the level of change associated with *Acacia* invasion. *Acacia* are dominated by a small number of species but are used opportunistically by many other species. Although the most common bird species detected where also urban-adapted birds, suggesting that these species do well in a variety of transformed landscapes, all species recorded in this study were native species.

The species richness and density of birds found in *Acacia* stands suggests that this habitat supports a large biomass of local species and, as a prominent landscape feature in the Western Cape, may be facilitating movement of species at landscape scales. Dures and Cumming (2010) found that *Acacia* presence in non-urban areas was associated with a reduction in species richness of Lowland Sand Fynbos bird communities. They concluded that *Acacia* presence was negatively affecting richness at the patch scale outside of urban areas but not influencing richness within urban areas where native vegetation is confined to more highly managed reserves. While species richness may decline outside of urban areas, the general conclusion that *Acacia* are a driver of species loss in this area seems overly simple (MacDougall and Turkington, 2005). In the CFR, agricultural areas cover 40% of lowland areas and is the leading cause of landscape transformation outside of the urban areas (Rouget *et al.*, 2003; Holmes *et al.* 2008). In some cases *Acacia* thickets may be purposely left on private farmland land to provide visual barriers, windbreaks, shade and supplementary fodder for animals in the summer (P. Stoffberg, J. De Flamingh personal communication), providing the only alternative habitat to agricultural fields. While *Acacia* do not support rare or specialist native bird species, *Acacia* presence in agricultural areas creates higher habitat complexity than if they were absent. While I observed birds perform territorial displays, feeding, and nesting attempts, further study of the reproductive success of birds nesting in *Acacia* will contribute greatly to the assessment of the quality of this habitat (Rodewald, 2011). However, in highly transformed landscapes, rather than reducing species richness, *Acacia* is likely increasing bird species richness, density and biomass. Furthermore, *Acacia* also provide space for other wild animals such as duiker, steenbok and caracal, all of which were found in many of the study sites (personal observation). The results of this study show that *Acacia* thickets can support high number and species of birds, however future studies that

wish to assess or compare the value of *Acacia* habitat to other habitat types should quantify population dynamics within *Acacia* and other habitats to fully understand the mechanisms underlying the patterns found here (Rodewald *et al.* 2011).

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Chapter 3 Conclusion

The significance of novel ecosystems for indigenous species and assemblages is poorly understood despite their growing presence at landscape and global scales (Hobbs *et al.* 2009). The formation of novel ecosystems presents a particular challenge for conservation as altered biotic structure and different abiotic processes support new combinations of species and species abundances (King and Hobbs, 2006; Lindenmayer *et al.* 2008). These habitats can create important habitat space for and support high numbers of indigenous species (Sogge *et al.* 2008; Quine and Humphrey, 2010). Despite this, they are often assumed to have low ecological value for native species (Davis *et al.* 2011; Rodewald, 2011). In South Africa, as elsewhere, the current understanding of community assembly in altered habitats is insufficient to predict which species will be supported by novel systems (Lindenmayer *et al.* 2008). Moreover, determining the significance of such habitats requires context-specific studies that evaluate both the negative and beneficial impacts on local biodiversity (Rodewald *et al.* 2011). While the negative impacts of invasive Australian *Acacia* species are well documented in the Cape Floristic Region (Richardson and van Wilgen, 2004; Yelenik *et al.* 2004; Seppelt *et al.* 2011), this study contributes to the growing body of work on novel ecosystems by examining the significance of *Acacia* thickets as habitat space for birds. In this chapter, I review key findings of my study questions, discuss important implications for management, and suggest important areas where further research is needed.

In Chapter 2, I first examine bird richness and abundance of assemblages within this novel habitat in the south-western Western Cape. The species supported by *Acacia* thickets are a subset of the avifauna in the region, and many of the endemic, habitat specialists are missing from *Acacia* assemblages. However, the high abundance of birds observed in this study suggests that acacias provide a large amount of habitat space for these species. Furthermore, the high species richness observed across study sites suggest that *Acacia* thickets may be used by many species as temporary habitat, and such habitat is highly valuable for dispersing individuals, especially in transformed landscapes (Penteriani *et al.* 2011).

Richness and abundance were evaluated across the landscape and found to be highly variable across sites. As found in other studies, variation in habitat structure and invasive plant cover mostly explained differences in bird assemblage richness and density between sites (Rotenberry, 1985; Armstrong and Van Hensbergen, 1994; Dures and Cumming, 2010). Both species richness and density decreased with increasing *Acacia* height and canopy

density. Therefore, bird assemblages will likely benefit from management and restoration efforts which increase or maintain shorter, indigenous, shrub vegetation.

Bird body size and feeding guilds were evaluated within *Acacia* assemblages to examine which birds are most likely to benefit from *Acacia* invasion. The bird assemblages within *Acacia* contain a moderate subset of the functional diversity of the regional bird species pool, although the species that are most abundant are small-bodied, foliage-gleaning, insectivores and mixed feeders. However, *Acacia* do not support all species within functional guilds equally and, because of species-specific habitat requirements, some common insectivorous species are absent from *Acacia* thickets (Fraser and Crowe, 1990; Winter and Faaborg, 2010). Additionally, the low abundance of some functional groups likely has consequences for ecological functioning (Wiens, 1989; Mayfield *et al.* 2010), and is worth further investigation. Many of the species that dominate *Acacia* stands are also urban adapted, suggesting these species possess traits that allow a common response to habitat disturbance (Henle *et al.* 2004; Devictor and Julliard, 2008). The dominance of a few species that are common across disturbed habitats, suggests that *Acacia* invasion, like other disturbance types, has a homogenizing effect on faunal communities at landscape scales (Devictor *et al.* 2007; Rooney *et al.* 2007; Dures and Cumming, 2010).

Using data from this study on bird abundance and body size, I provide an estimated range for the total population of birds and the total biomass supported by all *Acacia* thickets in the CFR. The average density estimate predicts that *Acacia* support 21 million birds with a biomass of 41 thousand kg. *Acacia* support a higher abundance of birds than some native vegetation types such as Restionaceous Tussock Marsh (Fraser and Crowe, 1990) but far lower abundances of birds than others such as Coastal Strandveld (Fox and Hockey, 2007) and Mountain Fynbos (Greve, 2006). However, given the total area invaded, and the high extent of habitat transformation in the Western Cape, *Acacia* provide a significant amount of habitat space for a large population of birds.

The results here provide base-line data on bird richness, abundance and the functional composition that contributes to our understanding of how avian assemblages are structured in a novel habitat. Furthermore, this study begins to explain why assemblage characteristics vary across habitat patches. I show that *Acacia* thickets in the south-western CFR do not support all species or all functional groups and contribute to homogenization of the regional bird fauna. However, *Acacia* do support a high abundance of birds, a high biomass of birds, provide a vast amount of habitat space, and support many other taxa. Management of areas invaded by *Acacia* that wish to support local bird assemblages should put a low priority on *Acacia* removal in highly transformed habitats. Additionally, restoration efforts that clear-cut

Acacia will benefit bird assemblages by maintaining habitat structure by preserving or re-introducing indigenous shrubs. Finally, *Acacia* thickets, in transformed areas should be viewed as contributing valuable structural diversity to habitat space and protected rather than neglected and treated as “trash” habitat. During the course of my research, further research questions became apparent that should be addressed to more fully understand bird use of novel *Acacia* habitat. How species specifically make use of *Acacia* habitat will require studies on nesting success (James, 1971), predation pressure (Schmidt and Whelan, 1999; Rodewald *et al.* 2011) and bird mobility through *Acacia* thickets (Winker *et al.* 1995). Additionally, future studies should compare assemblages within *Acacia* to those in other transformed habitats and investigate how *Acacia* thickets contribute to landscape-level habitat connectivity (Radford *et al.* 2005; Watson *et al.* 2005).

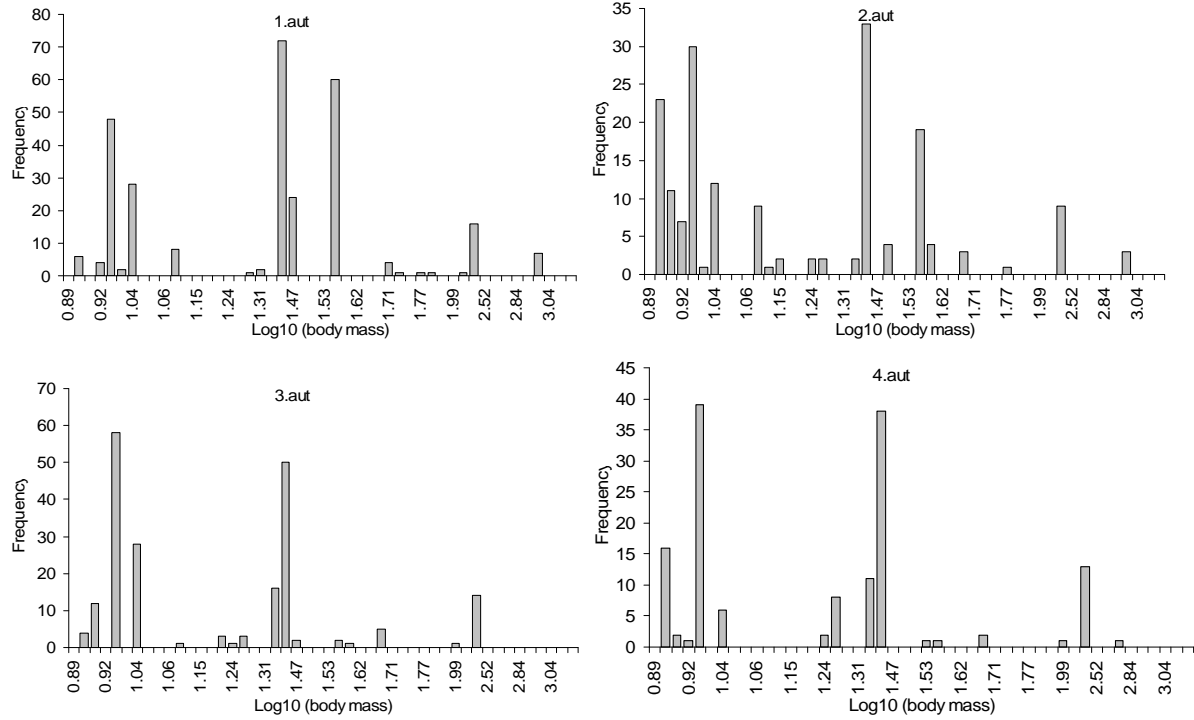
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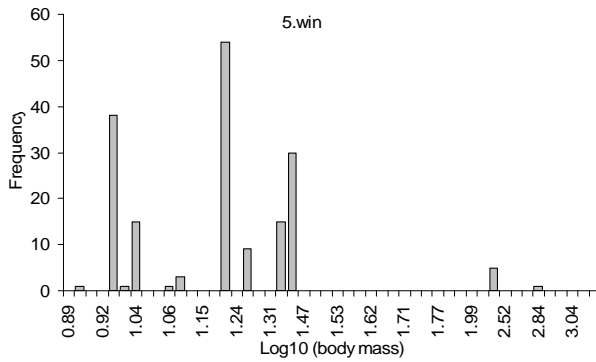
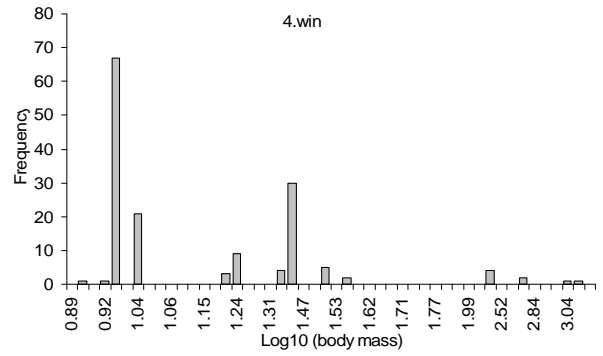
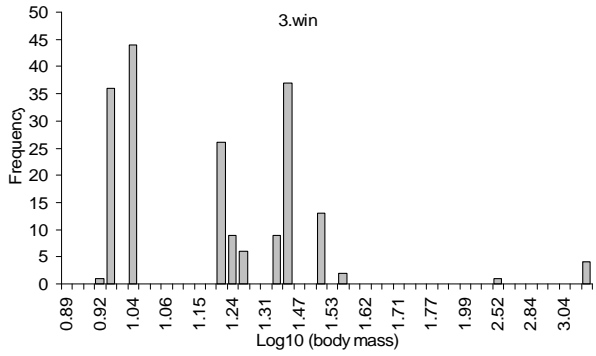
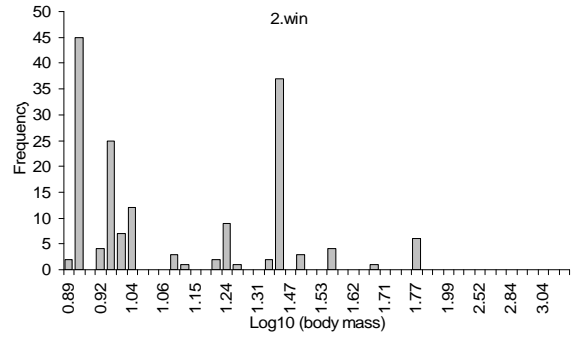
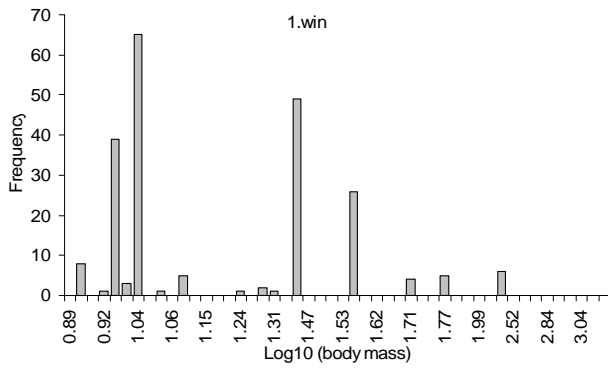
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Appendix I. Bird observation data

Figure 1. Body Size Frequency Distributions for each assemblage per survey. Titles are site number plus season abbreviation (aut = autumn, win = winter, spr = spring). Grey bars are the number of individuals observed for each body mass. Species appropriate mean body mass, obtained from literature, was assigned to each individual. Individuals were included here if observed within an area with a 45 m radius, during point counts.





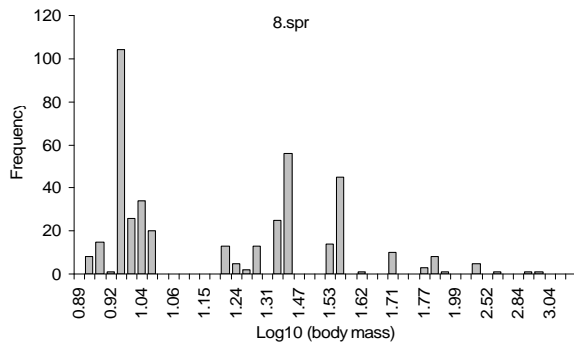
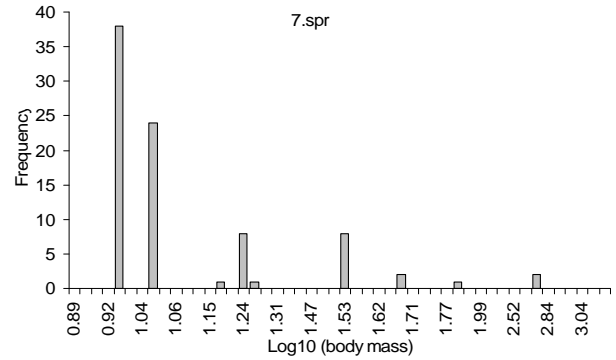
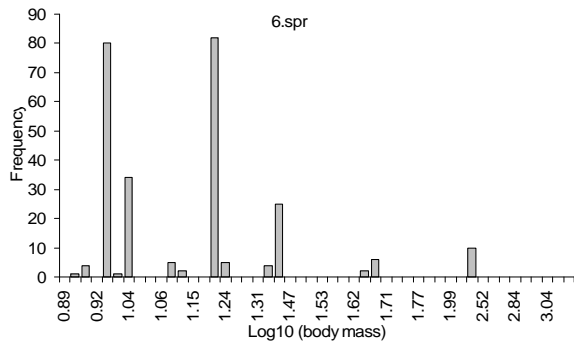
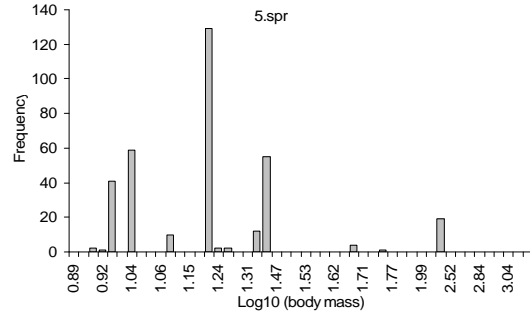
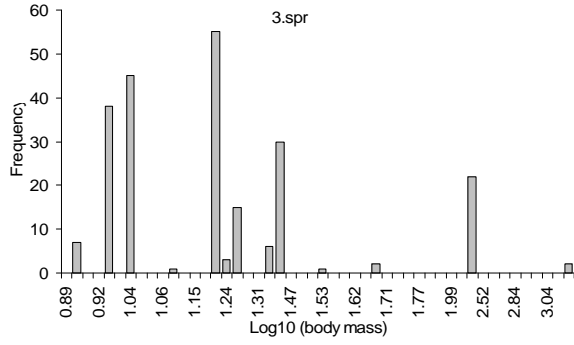
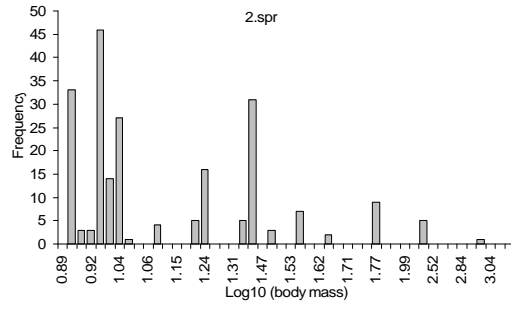
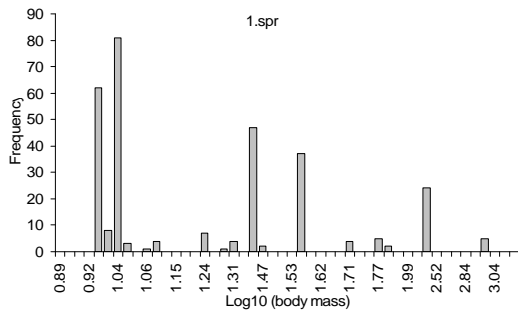


Table 1. Species list for all birds seen in *Acacia* listed by species number from Roberts Birds of Southern Africa 7th edition (2007).

<u>Roberts 7 #</u>	<u>Common name</u>	<u>Latin name</u>	<u>Mass (g)</u>
63	Heron Black-Headed	<i>Ardea melanocephala</i>	1440
91	Ibis Sacred	<i>Threskiornis aethiopicus</i>	1500
94	Ibis Hadedda	<i>Bostrychia hagedash</i>	1280
102	Goose Egyptian	<i>Alopochen aegyptiaca</i>	2110
116	Goose Spurwing	<i>Plectropterus gambensis</i>	4320
126	Kite Yellow-Billed	<i>Milvus parasitus</i>	828
127	Kite Black-Shouldered	<i>Elanus caeruleus</i>	247
136	Eagle Booted	<i>Aquila pennatus</i>	842
149	Buzzard Steppe	<i>Buteo vulpinus</i>	737
152	Buzzard Jackal	<i>Buteo rufofuscus</i>	1165
158	Sparrowhawk Black	<i>Accipiter melanoleucus</i>	725
160	Goshawk African	<i>Accipiter tachiro</i>	289
165	Marsh-Harrier African	<i>Circus ranivorus</i>	500
168	Harrier Black	<i>Circus maurus</i>	465
169	Harrier-Hawk African	<i>Polyboroides typus</i>	774
181	Kestrel Rock	<i>Falco naumanni</i>	216
195	Francolin Cape	<i>Pternistes capensis</i>	872
203	Guineafowl Helmeted	<i>Numida meleagris</i>	1440
208	Crane Blue	<i>Anthropoides paradiseus</i>	4870
258	Lapwing Blacksmith	<i>Vanellus armatus</i>	165
349	Pigeon Speckled	<i>Columba guinea</i>	344
350	Pigeon African Olive	<i>Columba arquatrix</i>	407
352	Dove Red-Eyed	<i>Streptopelia semitorquata</i>	235
354	Dove Cape Turtle	<i>Streptopelia capicola</i>	152
355	Dove Laughing	<i>Streptopelia senegalensis</i>	98
356	Dove Namaqua	<i>Oena capensis</i>	39
392	Owl Barn	<i>Tyto alba</i>	334
400	Eagle-Owl Cape	<i>Bubo capensis</i>	1139
401	Eagle-Owl Spotted	<i>Bubo africanus</i>	696
405	Nightjar Fiery-Necked	<i>Caprimulgus pectoralis</i>	57
424	Mousebird Speckled	<i>Colius striatus</i>	51
425	Mousebird White-Backed	<i>Colius colius</i>	42
426	Mousebird Red-Faced	<i>Urocolius indicus</i>	56
451	Hoopoe African	<i>Upupa africana</i>	53
518	Swallow Barn	<i>Hirundo rustica</i>	20
541	Drongo Fork-Tailed	<i>Dicrurus adsimilis</i>	43
547	Crow Cape	<i>Corvus capensis</i>	537
548	Crow Pied	<i>Corvus albus</i>	534
566	Bulbul Cape	<i>Pycnonotus capensis</i>	38
577	Thrush Olive	<i>Turdus olivaceus</i>	66
596	Stonechat African	<i>Saxicola torquatus</i>	14
601	Robin-Chat Cape	<i>Cossypha caffra</i>	28
614	Scrub-Robin Karoo	<i>Cercotrichas coryphoeus</i>	20
645	Apalis Bar-Throated	<i>Apalis thoracica</i>	11
651	Crombec Long-Billed	<i>Sylvietta rufescens</i>	12
661	Grassbird Cape	<i>Sphenoeacus afer</i>	31
669	Cisticola Grey-Backed	<i>Cisticola subruficapilla</i>	11
677	Cisticola Levallants	<i>Cisticola tinniens</i>	11
681	Neddicky	<i>Cisticola fulvicapilla</i>	8
686	Prinia Karoo	<i>Prinia maculosa</i>	9
690	Flycatcher Dusky	<i>Muscicapa adusta</i>	14
698	Flycatcher Fiscal	<i>Sigelus silens</i>	27
700	Batis Cape	<i>Batis capensis</i>	12

710	Flycatcher African Paradise	<i>Terpsiphone viridis</i>	14
713	Wagtail Cape	<i>Motacilla capensis</i>	21
716	Pipit African	<i>Anthus cinnamomeus</i>	25
727	Longclaw Cape	<i>Macronyx capensis</i>	48
732	Fiscal Common	<i>Lanius collaris</i>	40
736	Boubou Southern	<i>Laniarius ferrugineus</i>	59
746	Bokmakierie	<i>Telophorus zeylonus</i>	63
759	Starling Pied	<i>Spreo bicolor</i>	104
769	Starling Red-Winged	<i>Onychognathus morio</i>	135
775	Sunbird Malachite	<i>Nectarinia famosa</i>	17
783	Sunbird Lesser Double-Collared	<i>Cinnyris chalybeus</i>	8
796	White-Eye Cape	<i>Zosterops capensis</i>	11
801	Sparrow House	<i>Passer domesticus</i>	26
803	Sparrow Cape	<i>Passer melanurus</i>	30
813	Weaver Cape	<i>Ploceus capensis</i>	45
814	Weaver Southern Masked	<i>Ploceus velatus</i>	34
827	Bishop Yellow	<i>Euplectes capensis</i>	34
846	Waxbill Common	<i>Estrilda astrild</i>	8
850	Waxbill Swee	<i>Estrilda melanotis</i>	8
860	Whydah Pin-Tailed	<i>Vidua macroura</i>	15
872	Bunting Cape	<i>Emberiza capensis</i>	20
872	Canary Cape	<i>Serinus canicollis</i>	15
877	Canary Bully	<i>Serinus sulphuratus</i>	18

Table 2. Bird density data for each survey. Species abundances for each point are the pooled abundances from the six visits.

Koeberg NR Autumn				
Species	Distance	Individuals		
Apalis Bar-Throated	2	1	Mousebird Red-Faced	40
	40	1		Flyover
Batis Cape	10	1	Mousebird Speckled	28
	20	3		30
	30	3		40
	40	1		50
	55	5		52
	Flyover	2	Neddicky	19
Bokmakierie	18	1		40
Boubou Southern	40	1		55
	55	1	Prinia Karoo	5
Bulbul Cape	Flyover	1		6
	5	1		8
	10	2		10
	20	6		11
	30	28		17
	31	1		20
	32	1		22
	33	1		30
	39	3		36
	40	17		40
	50	1		41
	51	2		48
	55	35		50
	60	1		55
	65	3		Flyover
	Flyover	3	Robin-Chat Cape	5
Bunting Cape	40	2		8
	55	3		10
	Flyover	1		12
Buzzard Steppe	Flyover	4		13
Crow Pied	Flyover	7		17
Dove Cape Turtle	5	1		20
	10	4		22
	20	2		26
	21	3		27
	30	2		28
	40	3		29
	54	2		30
	55	7		32
	61	1		37
	64	2		40
	2	1		53
Flyover	2		55	
			Flyover	
Eagle-Owl Spotted	Flyover	2	Scrub-Robin Karoo	7
Fiscal Common	Flyover	1	Sparrow Cape	39
Francolin Cape	12	1		47
	30	2		52
	41	4		55
	55	2		Flyover
Heron Black-Headed	Flyover	1	Starling Red-Winged	10
Kite Black-Shouldered	Flyover	6	Sunbird Lesser Double-Collared	5

	10	1
	30	3
	34	1
	55	2
	Flyover	2
Waxbill Common	55	3
White-Eye Cape	10	1
	17	1
	18	1
	20	4
	30	12
	32	2
	40	7
	55	14
	Flyover	2

Roosands NR Autumn		
species	distance	Individuals
Apalis Bar-Throated	7	1
	55	1
Batis Cape	7	2
	30	2
	40	5
	55	10
Boubou Southern	40	1
	55	1
	Flyover	1
Bulbul Cape	9	1
	10	1
	22	2
	30	7
	32	1
	40	7
55	13	
Canary Cape	55	1
	Flyover	1
Dove Cape Turtle	30	4
	40	5
	55	3
Fiscal Common	30	1
	40	3
	55	1
Flycatcher African Paradise	7	1
	19	1
Flycatcher Dusky	30	1
Flycatcher Fiscal	28	1
	40	1
	Flyover	1
Francolin Cape	30	2
	40	1
	55	5
Goose Egyptian	Flyover	1
Grassbird Cape	10	1
	30	2
	40	1

	55	3
	Flyover	1
Guineafowl Helmeted	55	2
Heron Black-Headed	Flyover	1
Neddicky	13	1
	18	1
	20	1
	30	4
	55	2
Prinia Karoo	9	1
	10	1
	12	1
	13	1
	20	2
	27	2
	30	11
	40	11
	55	12
	Robin-Chat Cape	3
5		1
9		2
10		2
15		1
20		7
30		7
40		12
55	19	
Sunbird Lesser Double-Collared	5	1
	16	1
	20	8
	22	1
	28	1
	30	5
	40	6
	55	5
	Flyover	2
	Sunbird Malachite	30
Waxbill Common	20	3
	30	3
	32	2
	40	3
	55	3
Flyover	2	
Weaver Cape	30	2
	36	1
	56	1
White-Eye Cape	18	2
	20	1
	30	6
	40	3
55	3	
Canary Brimstone	40	2
	55	2

Penhill Estates Autumn

Species	Distance	Individuals
Batis Cape	20	1
	55	3
Bulbul Cape	30	1
	38	1
	52	1
	55	2
	Flyover	3
Canary Cape	10	3
Crow Pied	Flyover	4
Dove Cape Turtle	10	1
	11	1
	20	1
	30	6
	40	3
	45	2
	48	2
	50	1
	55	14
	Flyover	2
Dove Laughing	30	1
Dove Namaqua	Flyover	1
Eagle-Owl Spotted	Flyover	6
Fiscal Common	44	1
Flycatcher Fiscal	11	1
	13	1
	15	2
	20	2
	27	3
	30	1
	31	2
	32	1
	33	1
	40	1
	45	1
	55	2
	Flyover	1
	Francolin Cape	48
Goose Egyptian	Flyover	1
Guineafowl Helmeted	55	9
Heron Black-Headed	Flyover	2
Prinia Karoo	4	2
	5	2
	9	4
	10	2
	12	1
	14	1
	18	2
	20	12
	23	4
	27	1
	28	1
	29	1
	30	12
	32	1
40	12	

Robin-Chat Cape	55	14
	8	1
	10	4
	11	2
	18	1
	20	5
	28	2
	30	15
	40	20
	55	28
Sparrow Cape	57	1
	35	2
Sparrowhawk Black	Flyover	2
Sunbird Lesser Double-Collared	25	1
	30	1
	40	2
	Flyover	2
Sunbird Malachite	16	1
Thrush Olive	55	1
	Flyover	1
Waxbill Common	7	3
	8	6
	30	2
	40	1
	55	6
Weaver Cape	32	3
	35	1
	38	1
	52	1
	Flyover	2
White-Eye Cape	6	1
	15	1
	20	5
	27	1
	30	7
	38	2
	40	11
	55	21
	Flyover	1
	Canary Brimstone	20
40		2
55		1
60		5
Flyover		1

Rouxville Neighborhood Autumn		
Species	Distance	Individuals
Bishop Yellow	30	1
	55	5
	Flyover	2
Bulbul Cape	30	1
	55	2
Buzzard Jackal	Flyover	4
Canary Cape	Flyover	2

Crow Pied	30	1
	Flyover	5
Dove Cape Turtle	5	1
	10	3
	20	1
	28	1
	30	5
	40	2
	55	7
	58	1
Flyover	4	
Dove Laughing	20	1
Dove Namaqua	Flyover	1
Dove Red-Eyed	Flyover	1
Fiscal Common	48	1
	55	1
Flycatcher Fiscal	10	1
	12	3
	13	2
	20	2
	29	2
	30	1
	48	2
	50	4
	51	1
	55	4
Flyover	1	
Guineafowl Helmeted	55	11
Heron Black-Headed	Flyover	2
Kestrel Rock	Flyover	1
Kite Black-Shouldered	Flyover	2
Neddicky	2	1
	55	1
Prinia Karoo	10	5
	12	1
	15	2
	17	1
	20	8
	30	15
	40	7
	55	10
Flyover	2	
Robin-Chat Cape	2	1
	4	1
	5	1
	10	4
	16	1
	20	11
	30	8
	40	10
	55	9
	2	1
Sparrow Cape	50	1
	64	8
Sunbird Lesser Double-Collared	5	1

	8	1
	14	1
	17	1
	20	3
	30	7
	40	2
	55	3
Flyover	3	
Sunbird Malachite	10	1
	40	1
Flyover	2	
Swallow Barn	Flyover	5
Waxbill Common	20	2
	55	3
Weaver Cape	38	2
	Flyover	1
White-Eye Cape	20	4
	30	1
	34	1
	55	4
Canary Brimstone	30	2
	40	6
	55	4
	58	1

Koeberg NR Winter		
Species	Distance	Individuals
Apalis Bar-Throated	6	1
	30	2
	55	1
Batis Cape	20	1
	23	2
	30	1
	40	1
	55	9
Bokmakierie	55	2
	Flyover	2
Boubou Southern	16	1
	30	3
	40	1
	55	3
	Flyover	3
Bulbul Cape	10	1
	17	1
	19	2
	20	4
	30	5
	40	13
	47	1
	55	20
	Flyover	7
	Bunting Cape	32
55		1
Cisticola Levallants	8	1
Crow Pied	Flyover	7
Dove Cape Turtle	4	1
	6	1

	19	1
	20	2
	30	1
	55	1
	Flyover	2
Francolin Cape	48	2
	55	3
	64	2
	Flyover	1
Harrier Black	Flyover	4
Kestrel Rock	Flyover	2
Kite Black-Shouldered	Flyover	3
Mousebird Speckled	11	3
	23	1
	55	2
	Flyover	1
Neddicky	20	1
	55	1
Nightjar Fiery-Necked	Flyover	1
Prinia Karoo	3	1
	4	1
	5	2
	8	1
	10	3
	18	2
	19	1
	20	5
	22	1
	30	12
	38	1
	40	9
	55	15
Robin-Chat Cape	20	7
	30	18
	32	1
	36	1
	40	22
	55	39
	57	2
Scrub-Robin Karoo	20	1
	31	1
	Flyover	3
Sparrow Cape	55	2
	Flyover	1
Starling Pied	Flyover	2
Sunbird Lesser Double-Collared	4	1
	20	2
	40	5
	55	5
	Flyover	4
Sunbird Malachite	25	1
Weaver Cape	Flyover	1
White-Eye Cape	4	1
	5	1
	6	3

	10	1
	19	8
	20	3
	22	1
	24	1
	26	2
	27	4
	28	4
	29	3
	30	14
	32	2
	34	1
	38	1
	40	15
	55	13
	Flyover	17

Rooisands NR Winter		
Species	Distance	Individuals
Apalis Bar-Throated	16	1
	30	3
	40	3
	55	10
	Flyover	2
Batis Cape	7	1
	34	1
	40	1
	55	6
Boubou Southern	30	2
	40	4
	55	8
	Flyover	1
Bulbul Cape	10	1
	15	1
	36	1
	40	1
	55	2
	Flyover	1
Canary Cape	40	2
	Flyover	1
Dove Cape Turtle	Flyover	2
Fiscal Common	Flyover	1
Flycatcher Dusky	40	1
Flycatcher Fiscal	30	1
	34	1
	Flyover	2
Francolin Cape	55	3
Goose Egyptian	Flyover	2
Grassbird Cape	40	3
	55	3
Hoopoe African	Flyover	3
Ibis Hadeda	Flyover	2
Marsh-Harrier African	Flyover	1
Neddicky	30	2
	40	2
	55	1

Prinia Karoo	11	1
	20	2
	24	1
	30	9
	32	2
	40	10
	55	12
Robin-Chat Cape	6	1
	10	2
	14	1
	20	5
	30	13
	55	15
Sunbird Lesser Double-Collared	9	1
	11	1
	20	8
	30	14
	40	21
	55	28
	Flyover	2
Sunbird Malachite	14	1
	29	2
	40	6
	55	9
Thrush Olive	Flyover	1
Waxbill Common	Flyover	2
Waxbill Swee	22	2
Weaver Cape	40	1
	55	1
White-Eye Cape	12	1
	30	7
	40	4
	Flyover	2
Canary Brimstone	40	1
	55	2

Penhill Estates Winter		
Species	Distance	Individual
Batis Cape	55	3
Bishop Yellow	59	1
Bulbul Cape	40	2
	Flyover	1
Buzzard Jackal	Flyover	1
Canary Cape	18	4
	20	5
	27	1
	30	3
	40	13
	55	14
	Flyover	2
Crow Pied	Flyover	4
Dove Cape Turtle	55	2
Flycatcher Dusky	Flyover	1
Flycatcher Fiscal	22	1

	28	2
	30	2
	32	1
	40	2
	44	1
Guineafowl Helmeted	30	2
	40	2
	55	2
	Flyover	1
Kestrel Rock	Flyover	2
Neddicky	40	1
Owl Barn	12	1
Pigeon African Olive	55	1
Prinia Karoo	8	1
	14	1
	15	2
	18	4
	20	2
	22	2
	30	9
	40	15
55	21	
Robin-Chat Cape	7	2
	20	1
	30	20
	40	14
	55	26
	Flyover	1
Sparrow Cape	Flyover	2
Sunbird Lesser Double-Collared	55	2
	Flyover	2
Sunbird Malachite	10	2
	30	5
	40	2
	55	2
Waxbill Common	Flyover	1
Waxbill Swee	Flyover	2
Weaver Southern Masked	19	4
	20	1
	30	1
	32	1
	33	5
	35	1
	55	4
	59	3
Flyover	1	
White-Eye Cape	3	1
	7	1
	8	2
	9	5
	15	3
	20	4
	27	2
	30	8

	40	18
	50	5
	55	23
	Flyover	12
Canary Brimstone	19	2
	30	2
	40	2
	55	1

Rouxville Neighborhood Winter		
Species	Distance	Individuals
Bulbul Cape	22	2
	55	1
Buzzard Jackal	10	1
	Flyover	2
Canary Cape	40	3
	55	4
Crow Pied	10	1
	30	1
	Flyover	11
Dove Cape Turtle	5	1
	10	1
	20	1
	30	1
	55	4
	Flyover	3
Eagle-Owl Cape	28	1
Eagle-Owl Spotted	Flyover	1
Flycatcher Dusky	Flyover	1
Flycatcher Fiscal	14	1
	19	1
	33	1
	40	1
	55	1
	Flyover	1
Goose Egyptian	62	1
	Flyover	2
Guineafowl Helmeted	55	2
	Flyover	2
Heron Black-Headed	Flyover	2
Kite Black-Shouldered	Flyover	2
Neddicky	40	1
	55	1
Nightjar Fiery-Necked	56	1
Prinia Karoo	3	1
	7	1
	8	1
	10	3
	12	3
	13	2
	17	3
	20	5
	22	2
	26	1
	30	15
33	1	

	38	1
	40	28
	47	1
	55	27
	65	1
	Flyover	1
Robin-Chat Cape	10	2
	12	1
	13	1
	14	1
	20	2
	27	1
	30	11
	40	11
	55	18
	Flyover	1
Sunbird Lesser Double-Collared	30	1
	55	1
Sunbird Malachite	8	2
	10	1
	20	1
	22	1
	30	1
	40	3
	55	2
58	1	
	Flyover	1
Wagtail Cape	Flyover	1
Waxbill Common	55	2
Weaver Southern Masked	18	1
	19	1
	30	1
	33	2
	55	3
	Flyover	1
White-Eye Cape	9	2
	18	1
	20	1
	26	1
	30	10
	40	6
	55	10
	Flyover	5

Kerk te Koe Farm Winter		
Species	Distance	Individuals
Apalis Bar-Throated	20	1
	55	1
Batis Cape	8	1
	30	1
	40	1
	55	8
	Flyover	1
Buzzard Jackal	Flyover	1

Canary Cape	20	2
	21	1
	24	1
	30	20
	32	4
	40	26
	55	36
Crombec Long-Billed	9	1
Crow Pied	Flyover	6
Dove Cape Turtle	11	2
	38	2
	40	1
	55	1
	Flyover	4
Eagle-Owl Spotted	6	1
Fiscal Common	55	1
Flycatcher Fiscal	12	1
	18	1
	20	1
	30	4
	32	1
	34	1
	36	1
	38	3
	41	1
	42	1
	49	1
	52	1
55	4	
Goose Egyptian	Flyover	4
Goose Spurwing	Flyover	1
Heron Black-Headed	Flyover	1
Kestrel Rock	Flyover	1
Neddicky	55	1
Nightjar Fiery-Necked	Flyover	2
Prinia Karoo	4	1
	5	1
	7	1
	9	1
	10	1
	11	1
	13	2
	15	1
	18	1
	20	2
	30	9
	35	3
	40	14
	55	21
64	1	
Robin-Chat Cape	4	1
	8	1
	20	3
	24	1
	30	12
	35	1

	40	11
	55	11
	64	1
Sparrow Cape	55	1
Sunbird Lesser Double-Collared	40	1
Swallow Barn	Flyover	1
Weaver Cape	Flyover	1
White-Eye Cape	20	1
	30	1
	40	13
	55	10
	Flyover	13
Canary Brimstone	23	4
	30	2
	40	3
	55	6

Koeberg NR Spring		
Species	Distance	Individuals
Apalis Bar-Throated	15	2
	20	2
	30	2
	40	2
	55	7
	65	2
	Flyover	2
Batis Cape	2	1
	23	2
	30	1
	55	2
	65	11
Bokmakierie	38	1
	40	1
	65	4
	Flyover	1
Boubou Southern	10	1
	30	2
	40	2
	65	8
	Flyover	1
Bulbul Cape	2	1
	4	1
	8	2
	20	7
	25	1
	29	1
	30	9
	32	1
	36	3
	40	8
	43	3
	46	1
	52	1
55	11	
65	9	

	Flyover	2
Bunting Cape	20	1
	22	1
	27	1
	35	1
	55	1
	Flyover	3
Canary Cape	65	1
Cisticola Grey-Backed	20	1
	30	1
	40	1
	65	1
Crombec Long-Billed	34	1
	Flyover	1
Crow Pied	Flyover	19
Dove Cape Turtle	5	1
	8	1
	10	3
	13	1
	20	4
	30	8
	40	6
	52	1
	55	16
	61	1
	65	33
Flyover	1	
Eagle-Owl Spotted	Flyover	1
Francolin Cape	39	3
	40	1
	44	1
	55	3
	65	13
Marsh-Harrier African	Flyover	2
Mousebird Speckled	6	1
	10	1
	21	1
	30	1
Prinia Karoo	2	2
	8	2
	10	2
	13	1
	16	1
	18	2
	20	17
	26	2
	27	2
	28	1
	30	10
	32	1
	36	1
	40	17
	44	1
	48	1
	55	13
58	1	

	65	17
Robin-Chat Cape	5	1
	10	3
	20	12
	26	2
	27	2
	30	10
	32	1
	40	16
	46	1
	55	10
	60	1
Scrub-Robin Karoo	62	1
	65	10
	28	1
	50	1
	Flyover	1
Sparrow Cape	18	2
	Flyover	1
Starling Pied	Flyover	1
Sunbird Lesser Double-Collared	55	1
	Flyover	1
Sunbird Malachite	21	1
	27	2
	30	1
	40	3
	52	2
	65	1
	Flyover	2
Thrush Olive	55	1
White-Eye Cape	2	1
	8	3
	10	6
	16	2
	18	1
	19	1
	20	18
	30	28
	33	3
	40	18
	52	1
	55	12
	65	8
Flyover	5	

Rooisands NR Spring		
Species	Distance	Individuals
Apalis Bar-Throated	20	2
	30	4
	40	8
	55	10
	Flyover	1
Batis Cape	20	1
	30	1
	40	2
	55	5

	Flyover	2
Boubou Southern	20	3
	30	1
	40	5
	55	8
	Flyover	1
Bulbul Cape	30	2
	40	5
	55	1
	Flyover	3
Canary Cape	14	1
	20	2
	40	2
	Flyover	1
Cisticola Grey-Backed	40	1
	55	1
Dove Cape Turtle	20	1
	23	1
	30	1
	40	2
	55	6
	Flyover	1
Drongo Fork-Tailed	40	2
	55	1
Flycatcher Fiscal	17	1
	25	1
	34	1
	40	2
	50	1
	60	1
Francolin Cape	25	1
	Flyover	3
Goshawk African	Flyover	1
Grassbird Cape	30	1
	40	2
	55	3
	Flyover	1
Guineafowl Helmeted	Flyover	1
Kite Black-Shouldered	Flyover	1
Neddicky	30	1
	38	1
	40	1
	55	1
	Flyover	1
Prinia Karoo	7	1
	10	4
	12	1
	13	2
	18	2
	20	10
	22	1
	30	13
	40	12
	55	6
	60	1
Robin-Chat Cape	9	1

	18	1
	20	8
	30	11
	40	10
	55	16
Sunbird Lesser Double-Collared	10	2
	13	1
	20	8
	30	9
	35	1
	40	12
	Flyover	3
Sunbird Malachite	10	3
	20	3
	30	6
	40	4
	53	2
	55	5
Waxbill Common	20	2
	40	1
White-Eye Cape	8	1
	9	2
	10	4
	20	6
	22	1
	30	7
	40	6
	55	11
	Flyover	10

Penhill Estates Spring		
Species	Distance	Individuals
Batis Cape	30	1
	55	1
	65	4
Bishop Yellow	30	1
Canary Cape	10	1
	18	2
	19	1
	20	11
	25	2
	30	19
	36	2
	38	2
	40	15
	49	1
	50	1
	54	3
	55	22
	56	4
61	2	
65	35	
Flyover	4	
Crow Pied	Flyover	2

Dove Cape Turtle	11	1
	12	2
	18	1
	20	3
	27	1
	30	2
	32	1
	33	1
	36	2
	40	6
	42	2
	55	20
	65	41
Drongo Fork-Tailed	Flyover	1
Flycatcher Fiscal	9	1
	20	1
	28	1
	32	1
	40	2
	55	1
	61	1
	64	1
	65	1
	Flyover	1
Guineafowl Helmeted	20	1
	30	1
	65	3
		Flyover
Heron Black-Headed	65	1
		Flyover
Kite Black-Shouldered	Flyover	2
Kite Yellow-Billed	Flyover	2
Prinia Karoo	9	1
	20	6
	21	1
	30	14
	36	1
	40	15
	55	12
65	5	
Robin-Chat Cape	20	5
	30	11
	40	14
	55	14
	65	14
Sparrow Cape	Flyover	1
Sparrow House	55	1
Sunbird Lesser Double-Collared	9	2
	10	1
	20	1
	40	3
Sunbird Malachite	20	1
	30	1
	40	1
		Flyover

Waxbill Common	Flyover	5
Weaver Cape	40	2
White-Eye Cape	6	3
	7	1
	8	2
	9	2
	10	1
	11	1
	12	1
	13	2
	17	2
	18	1
	20	8
	24	1
	30	8
	34	2
	40	9
43	1	
55	3	
	Flyover	4
Whydah Pin-Tailed	Flyover	1
Canary Brimstone	9	1
	16	1
	20	1
	27	6
	30	1
	34	2
	40	3
	55	1
	65	1
		Flyover

Kerk te Koe Farm Spring		
Species	Distance	Individuals
Batis Cape	4	1
	6	1
	10	1
	30	2
	40	5
	55	1
		Flyover
Bishop Yellow	Flyover	2
Bokmakierie	Flyover	1
Buzzard Jackal	Flyover	4
Canary Cape	3	2
	6	1
	8	2
	10	3
	11	1
	12	1
	13	1
	16	3
	18	1
	20	31
23	2	
24	2	

	27	1
	29	1
	30	35
	31	3
	33	3
	40	36
	50	4
	53	2
	55	34
	Flyover	7
Crombec Long-Billed	Flyover	1
Crow Pied	Flyover	19
Dove Cape Turtle	19	1
	20	4
	30	9
	40	5
	55	23
	Flyover	4
Flycatcher Fiscal	7	2
	16	1
	20	2
	23	4
	30	1
	32	1
	45	1
	46	1
	55	1
	Flyover	1
Guineafowl Helmeted	55	2
	Flyover	1
Heron Black-Headed	Flyover	1
Kite Black-Shouldered	Flyover	3
Kite Yellow-Billed	Flyover	5
Neddicky	30	1
Nightjar Fiery-Necked	34	1
Prinia Karoo	4	1
	8	1
	10	1
	14	1
	15	1
	16	1
	17	1
	20	14
	24	1
	25	1
	30	6
	31	1
	32	1
	40	10
	55	9
	Flyover	1
Robin-Chat Cape	2	1
	7	1
	10	4
	12	1
	16	1

	18	1
	20	3
	21	1
	30	22
	40	20
	55	9
Sunbird Malachite	17	1
	30	1
	53	1
	55	1
	Flyover	1
Waxbill Common	25	1
	31	1
Weaver Cape	20	1
	30	1
	37	1
	40	1
	55	1
White-Eye Cape	3	2
	4	1
	7	4
	8	1
	9	3
	10	2
	11	3
	15	1
	16	1
	17	2
	18	1
	20	13
	21	1
	30	11
	32	2
	36	2
	40	9
	55	2
	Flyover	8
Canary Brimstone	20	1
	30	1

De Grendel Estates Spring		
Species	Distance	Individuals
Apalis Bar-Throated	40	1
	55	2
Batis Cape	15	1
	20	1
	30	1
	40	2
	55	3
	Flyover	2
Boubou Southern	Flyover	1
Buzzard Jackal	Flyover	2
Canary Cape	1	1
	4	1
	10	2

	11	2
	12	2
	18	2
	19	3
	20	12
	24	1
	30	19
	38	2
	40	35
	52	5
	55	22
	Flyover	5
Crow Pied	Flyover	7
Dove Cape Turtle	1	1
	30	3
	40	6
	55	4
	Flyover	1
Drongo Fork-Tailed	8	1
	13	1
	55	1
	Flyover	2
Fiscal Common	Flyover	1
Flycatcher Dusky	20	1
	32	1
Flycatcher Fiscal	24	4
	46	1
	57	1
	Flyover	5
Goshawk African	52	1
	Flyover	1
Guineafowl Helmeted	47	1
	55	1
	57	1
	Flyover	3
Harrier Black	Flyover	4
Kite Black-Shouldered	Flyover	1
Kite Yellow-Billed	Flyover	7
Prinia Karoo	3	1
	4	1
	6	1
	7	1
	8	1
	10	6
	13	2
	15	1
	16	1
	19	1
	20	16
	21	3
	22	1
	25	1
	26	2
	28	1
	30	20
	32	3

	33	2
	40	14
	44	1
	55	10
	Flyover	1
Robin-Chat Cape	10	1
	20	7
	24	1
	30	9
	40	7
	55	16
	Flyover	3
Sunbird Lesser Double-Collared	30	1
Sunbird Malachite	6	2
	12	2
	30	1
	55	1
Waxbill Common	9	2
	36	1
	39	1
	Flyover	3
Weaver Cape	1	1
	3	1
	22	1
	27	1
	32	2
	57	1
	Flyover	4
White-Eye Cape	6	3
	10	5
	11	2
	12	1
	13	1
	15	1
	17	1
	20	8
	27	2
	30	6
	38	1
	40	3
	55	4

Riverlands NR Spring		
Species	Distance	Individuals
Bishop Yellow	13	2
	25	1
	32	1
	37	1
	40	2
	42	1
	48	1
	55	7
	65	2
	Flyover	1
Bokmakierie	18	1

	55	8
	65	8
Bulbul Cape	55	1
Canary Cape	55	1
Cisticola Grey-Backed	3	2
	5	1
	10	3
	19	2
	26	1
	28	2
	30	1
	34	1
	35	1
	36	1
	40	8
	41	1
	46	2
	52	2
	55	11
	56	2
	65	4
Crane Blue	20	2
	Flyover	2
Crow Pied	20	2
	Flyover	4
Eagle Booted	Flyover	1
Flycatcher Fiscal	Flyover	1
Ibis Sacred	Flyover	1
Kite Yellow-Billed	Flyover	4
Longclaw Cape	55	1
	65	1
Pipit African	41	1
	52	1
	54	1
	55	1
	Flyover	1
Prinia Karoo	7	1
	18	1
	20	3
	22	1
	24	1
	26	1
	30	5
	31	1
	34	5
	40	12
	42	2
	43	3
	44	1
	45	1
	52	1
	53	1
	55	18
	56	1
	65	3
	Flyover	1

Robin-Chat Cape	55	1
	65	1
Stonechat African	20	1
	48	2
	Flyover	4
Sunbird Malachite	8	1
	14	1
	20	2
	26	1
	30	1
	35	1
	40	1
	55	4
	65	1
	Flyover	8
Waxbill Common	64	3
Weaver Cape	37	2
White-Eye Cape	Flyover	1
Canary Brimstone	40	1

Bottelary Conservancy Spring		
Species	Distance	Individuals
Apalis Bar-Throated	3	2
	8	2
	9	1
	10	2
	17	2
	20	2
	22	3
	26	1
	27	1
	30	5
	40	5
	55	6
	Flyover	1
Batis Cape	55	1
	Flyover	2
Bishop Yellow	10	1
	13	1
	18	1
	22	1
	30	1
	31	1
	33	1
	34	1
	37	2
	40	3
	43	1
	55	3
	Flyover	6
Bokmakierie	10	2
	12	2
	15	1
	30	3
	49	3
	55	2

	Flyover	2
Boubou Southern	33	2
	40	1
	55	1
	Flyover	3
Bulbul Cape	4	1
	7	1
	11	2
	12	1
	17	4
	18	3
	20	1
	21	2
	22	1
	23	1
	25	2
	26	3
	27	1
	28	1
	30	3
	31	2
	35	1
	37	1
	39	3
	40	10
43	1	
49	6	
55	13	
	Flyover	1
Buzzard Jackal	Flyover	2
Canary Cape	9	1
	14	1
	19	1
	20	1
	26	1
	30	2
	32	2
	35	2
	40	2
	49	1
	55	2
	Flyover	2
Cisticola Grey-Backed	5	1
	8	1
	10	1
	11	1
	13	2
	17	3
	20	4
	21	1
	22	1
	27	2
	32	1
	40	2
	55	1
60	1	

	Flyover	1
Crow Cape	Flyover	1
Crow Pied	Flyover	4
Dove Cape Turtle	20	1
	22	1
	27	1
	38	1
	40	1
	55	2
	Flyover	1
Drongo Fork-Tailed	Flyover	1
Eagle Booted	Flyover	3
Flycatcher Fiscal	6	1
	14	1
	16	2
	19	1
	20	1
	23	1
	24	3
	25	1
	26	1
	29	2
	30	2
	32	4
	35	1
	38	3
	43	1
46	1	
49	2	
55	1	
58	1	
	Flyover	4
Francolin Cape	8	1
	Flyover	4
Goshawk African	Flyover	1
Guineafowl Helmeted	55	1
	Flyover	3
Harrier Black	Flyover	1
Harrier-Hawk African	Flyover	1
Kite Yellow-Billed	27	1
	Flyover	4
Mousebird Red-Faced	Flyover	2
Mousebird Speckled	8	1
	15	1
	20	2
	27	1
	32	1
	34	4
	60	1
		Flyover
Mousebird White-Backed	8	1
	Flyover	1
Neddicky	40	1
	55	1
	Flyover	1

Pigeon Speckled	40	1
Prinia Karoo	5	2
	6	1
	7	1
	8	3
	9	1
	10	2
	11	1
	12	1
	15	2
	16	1
	18	2
	19	4
	20	14
	21	3
	22	6
	23	1
	25	1
	27	4
	28	1
	29	1
	30	23
	32	1
	33	1
34	1	
40	24	
41	2	
55	19	
58	2	
60	1	
Robin-Chat Cape	9	1
	10	3
	13	1
	14	1
	15	2
	16	1
	18	2
	19	1
	20	8
	22	1
	23	1
	25	2
	27	1
	28	1
	29	1
	30	10
	32	1
	36	1
	40	16
	43	1
48	2	
55	10	

	60	1
	Flyover	1
Scrub-Robin Karoo	8	1
	10	4
	12	1
	17	1
	20	1
	22	1
	23	2
	24	1
	28	1
	55	1
	60	1
	Flyover	3
Sunbird Lesser Double-Collared	10	1
	11	2
	13	1
	18	2
	19	1
	40	1
	Flyover	4
Sunbird Malachite	22	3
	30	2
Thrush Olive	28	1
Waxbill Common	13	4
	14	1
	15	1
	23	1
	31	5
	34	3
	47	1
	Flyover	8
Waxbill Swee	Flyover	1
Weaver Cape	Flyover	2
White-Eye Cape	8	2
	9	5
	13	3
	14	1
	16	1
	20	8
	28	2
	30	5
	34	1
	36	1
40	5	
55	5	
Flyover	3	
Whydah Pin-Tailed	Flyover	1
Canary Brimstone	22	2

Appendix II. Vegetation survey data

Figure 1: Percent ground cover for each site for the eight categories. Bar height is mean cover for each site, and error bars are standard error.

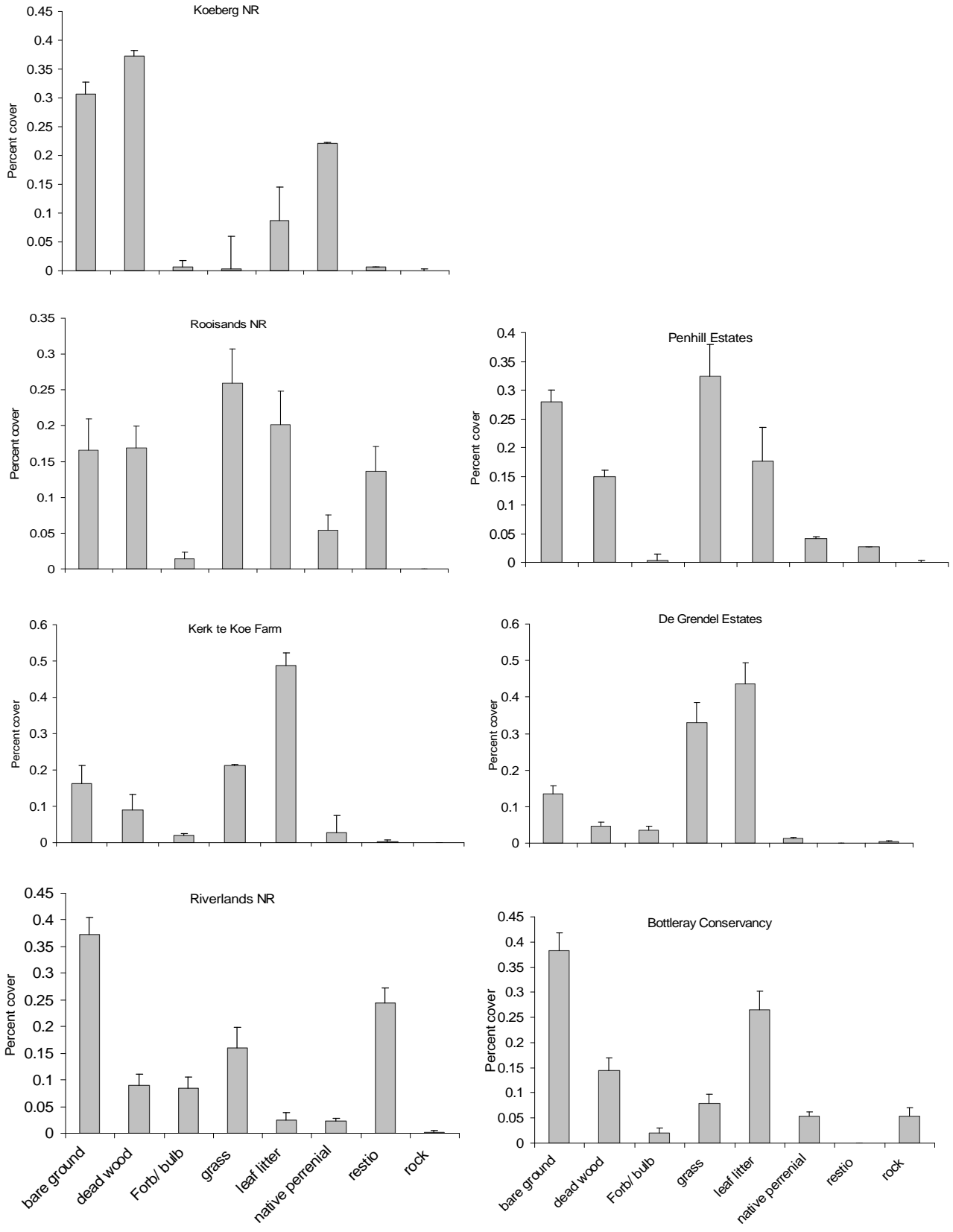


Figure 2. Vegetation profile histograms. The frequency that vegetation touched a pole (demarcated in 10 cm intervals) placed vertically in the vegetation. Vegetation structure varied across sites with dominant invasive species. Koeberg NR is invaded by *Acacia cyclops*. Rooisands NR is mixed *A. saligna* and *A. cyclops*. Penhill Estates is dominated by *A. saligna*. Kerk te Koe Farm is dominated by *A. saligna*. De Grendel Estates is dominated by *A. saligna*. Riverlands NR and Bottelary Conservancy are the least invaded, but the dominant invasive species is *A. saligna*.

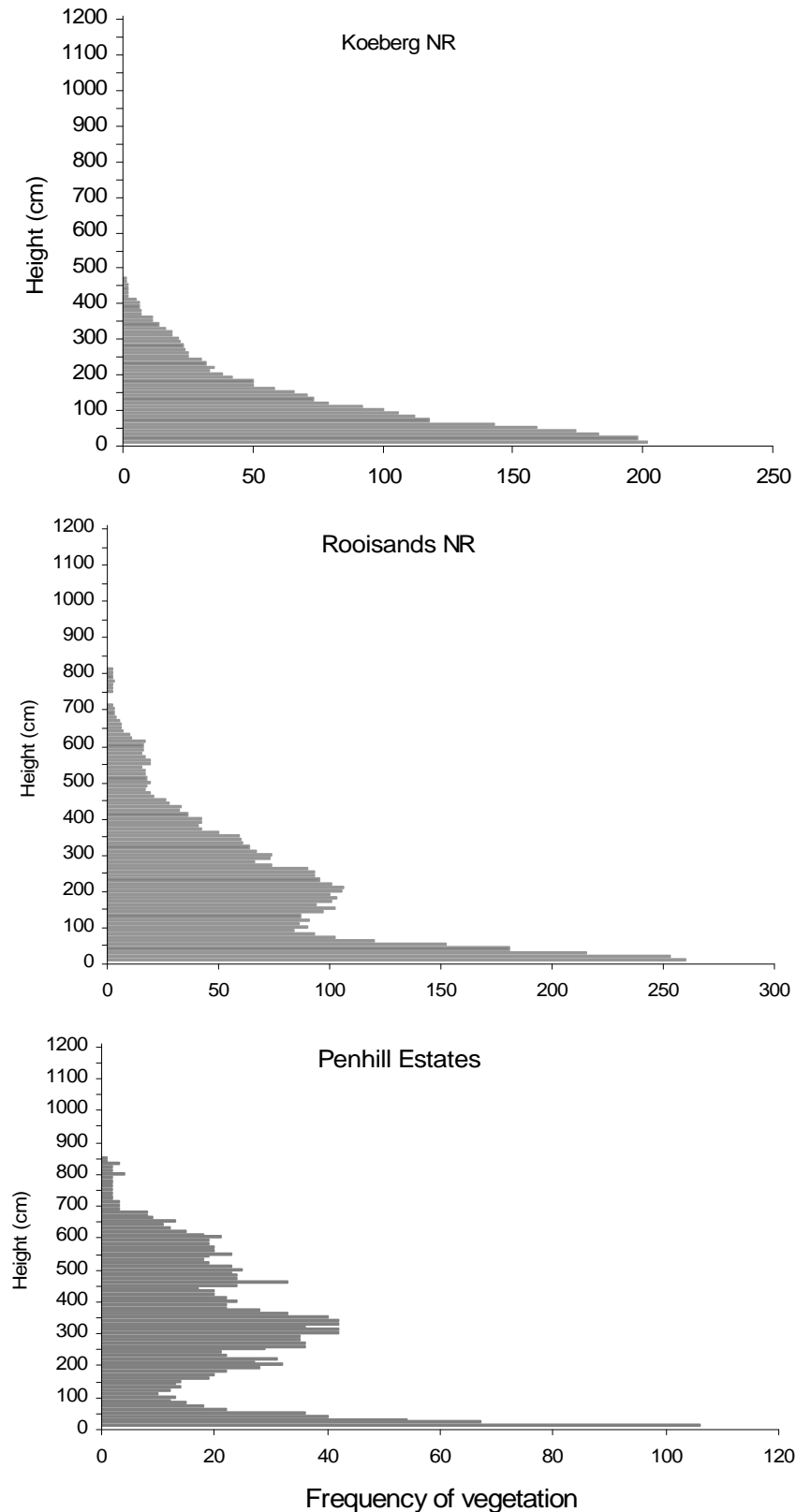


Figure 2. *Continued*

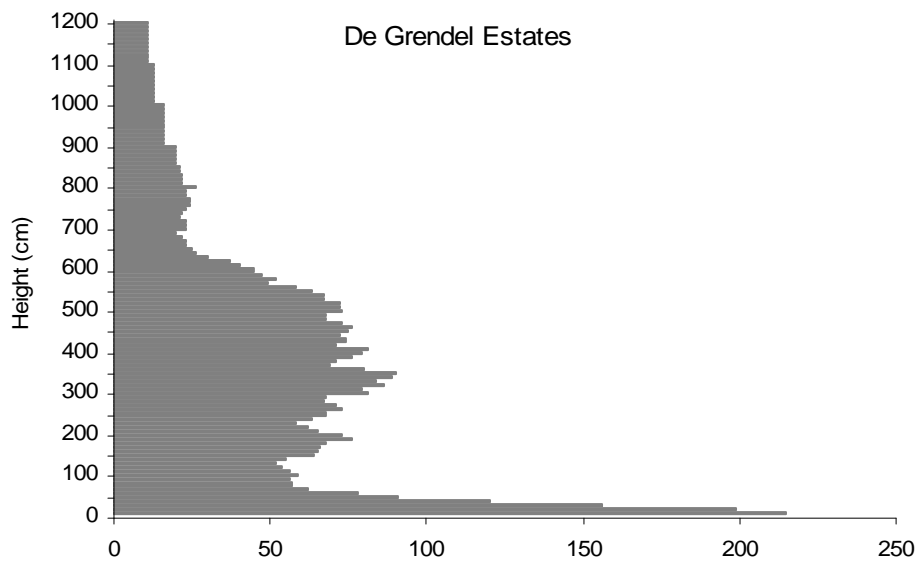
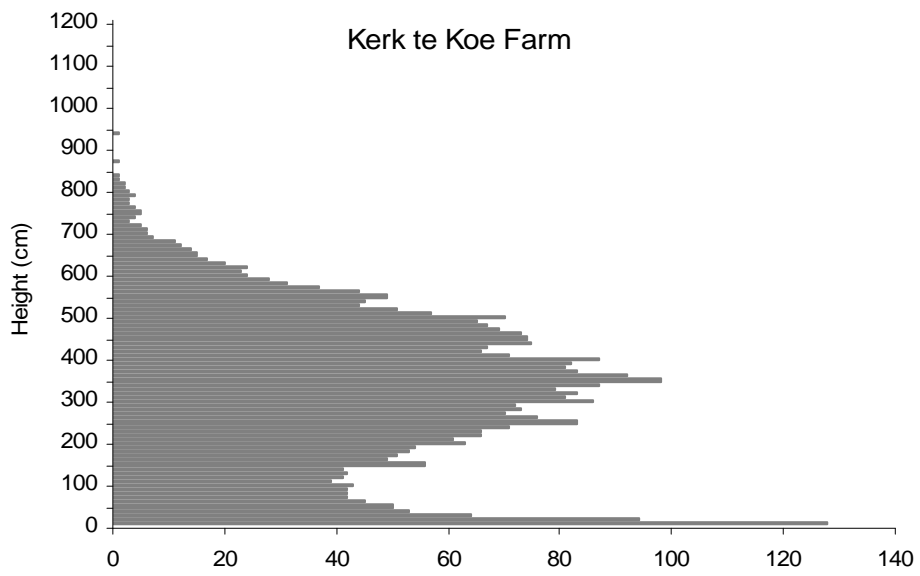


Figure 2. *Continued*

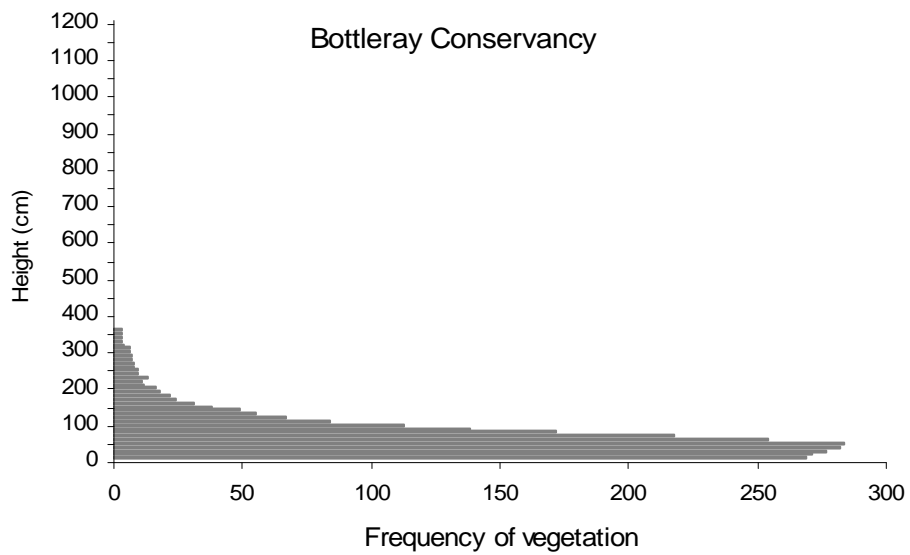
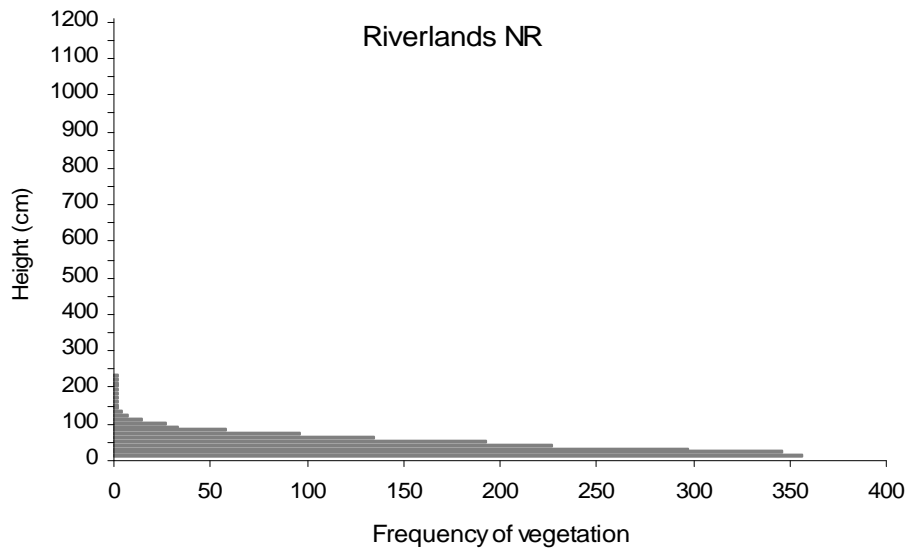


Table 1. Vegetation data per transect for each survey site. Canopy cover and ground cover types listed as percentages for each transect. Vegetation surveys were only conducted at points which had not experienced fire or wood cutting.

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %	
De Grendel	1	N	3.18	47	14	0	0	0	87	0	0	0	0	
		SE	1.72	34	7	7	0	0	87	0	0	0	0	
		SW	1.83	94	0	7	0	0	14	80	0	0	0	0
	2	N	0.2	14	34	0	20	47	0	0	0	0	0	0
		SE	0.72	27	0	0	0	87	7	7	7	0	0	0
		SW	0.22	14	0	7	0	87	7	0	0	0	0	0
	4	N	2.43	34	14	0	0	0	27	54	0	7	0	0
		SE	0.5	27	7	0	7	87	0	0	0	0	0	0
		SW	2.22	20	7	0	14	14	14	60	0	0	0	7
	5	N	3.63	7	7	0	0	0	54	34	0	7	0	0
		SE	0.52	20	0	14	20	67	67	0	0	0	0	0
		SW	1.97	40	20	7	0	0	0	74	0	0	0	0
	6	N	1.7	14	34	0	14	47	47	7	0	0	0	0
		SE	0	0	7	0	7	74	74	14	0	0	0	0
		SW	0.49	20	40	0	0	0	27	14	20	0	0	0
	7	N	0.72	14	27	0	7	67	67	0	0	0	0	0
		SE	1.25	34	20	0	14	67	67	0	0	0	0	0
		SW	1.17	40	0	0	7	74	74	20	0	0	0	0
	8	N	2.29	34	7	7	0	0	0	87	0	0	0	0
		SE	2.68	34	27	0	0	0	7	67	0	0	0	0
		SW	0.95	54	27	7	0	0	0	67	0	0	0	0
	9	N	0.97	40	14	20	0	0	7	60	0	0	0	0
		SE	1.52	34	0	7	0	0	20	74	0	0	0	0
		SW	5.06	47	7	20	0	0	34	40	0	0	0	0
	10	N	1.99	47	14	14	0	0	7	67	0	0	0	0
		SE	4.63	60	7	7	0	0	34	54	0	0	0	0
		SW	8.81	60	14	7	0	0	40	40	0	0	0	0
	11	N	3.58	40	7	0	0	0	0	94	0	0	0	0
		SE	3.08	34	27	7	0	0	0	60	0	0	0	7
		SW	0.24	60	27	7	0	0	7	60	0	0	0	0

Table 1. Continued

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %	
Koeberg	1	N	1	22	22	43	0	0	22	0	15	0	0	
		SE	0.93	0	22	29	0	0	36	0	15	0	0	
		SW	0	29	15	29	0	0	58	0	0	0	0	
	2	N	1.33	15	72	29	0	0	0	0	0	0	0	0
		SE	3.02	50	8	36	0	0	58	0	0	0	0	0
		SW	1.7	43	29	36	0	0	22	0	15	0	0	0
	3	N	0.35	8	58	36	0	0	0	0	0	8	0	0
		SE	1.04	22	29	72	0	0	0	0	0	0	0	0
		SW	0.87	58	72	29	0	0	0	0	0	0	0	0
	6	N	3.37	15	29	29	0	0	0	0	0	43	0	0
		SE	6.08	8	22	22	0	0	0	0	0	58	0	0
		SW	2.58	8	8	36	8	0	0	0	0	43	8	0
	7	N	0	0	100	0	0	0	0	0	0	0	0	0
		SE	0	0	50	0	8	0	0	0	0	43	0	0
		SW	0.52	0	43	43	0	0	0	0	0	15	0	0
	8	N	0.47	0	15	79	0	0	0	0	0	8	0	0
		SE	0.1	0	15	79	0	0	0	0	0	8	0	0
		SW	0.24	0	22	58	0	0	0	0	0	22	0	0
	9	N	0.58	15	29	22	0	0	0	0	0	50	0	0
		SE	0.68	0	22	50	0	0	0	15	0	15	0	0
		SW	0.77	0	8	22	0	0	0	0	0	72	0	0
10	N	0.22	8	50	43	0	0	0	0	0	8	0	0	
	SE	0.66	0	8	65	0	0	0	0	0	29	0	0	
	SW	3.1	22	0	15	0	0	8	0	0	72	8	0	

Table 1. Continued

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %
Melkbos	1	N	2.33	40	20	14	0	0	67	0	0	0	0
		SE	3.11	34	7	47	0	7	34	0	0	7	0
		SW	2.2	60	14	0	0	14	74	0	0	0	0
	2	N	3.5	34	14	14	0	0	74	0	0	0	0
		SE	1.43	47	27	14	0	0	40	0	20	0	0
		SW	2.74	40	14	0	0	0	74	0	14	0	0
	3	N	0.59	40	14	14	7	7	60	0	0	0	0
		SE	0.9	40	14	34	0	0	54	0	0	0	0
		SW	0.9	47	14	7	0	14	67	0	0	0	0
	4	N	1.17	54	27	7	0	14	54	0	0	0	0
		SE	2.63	54	7	34	0	0	60	0	0	0	0
		SW	0.13	7	34	7	0	34	27	0	0	0	0
	5	N	0.5	67	14	0	0	54	34	0	0	0	0
		SE	0	0	40	0	7	47	0	0	7	0	0
		SW	1.5	40	7	7	0	0	87	0	0	0	0
	6	N	4.33	60	0	0	0	0	100	0	0	0	0
		SE	6.63	47	0	7	0	0	87	0	7	0	0
		SW	4.27	60	7	14	0	0	80	0	0	0	0
	8	N	0.42	40	14	0	0	60	27	0	0	0	0
		SE	0.58	54	7	20	0	40	34	0	0	0	0
		SW	1.34	54	27	0	0	14	60	0	0	0	0
	9	N	0.08	34	7	0	0	87	7	0	0	0	0
		SE	0.42	20	20	0	0	74	0	7	0	0	0
		SW	1.43	40	27	7	0	0	67	0	0	0	0
	10	N	0.25	27	27	0	34	7	14	20	0	0	0
		SE	0.99	27	40	0	7	14	40	0	0	0	0
		SW	0.33	20	7	0	0	94	0	0	0	0	0

Table 1. Continued

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %	
Mooiplaas	1	N	0.33	15	29	0	0	22	15	0	15	0	22	
		SE	0.97	0	36	22	0	22	22	0	0	0	0	
		SW	0.41	0	36	15	0	22	29	0	0	0	0	
	2	N	0	0	54	0	0	0	0	14	0	0	0	34
		SE	1.27	0	54	0	0	0	0	40	0	0	0	7
		SW	0.15	0	67	0	0	0	0	14	7	0	0	14
	3	N	0.54	0	43	15	0	0	22	0	0	0	0	22
		SE	0.24	0	22	15	0	0	0	65	0	0	0	0
		SW	1.35	0	58	22	0	0	15	0	0	8	0	0
	4	N	3.37	0	8	15	0	0	0	65	0	8	0	8
		SE	0.79	72	8	36	0	0	22	29	0	0	0	8
		SW	0.35	8	22	50	0	0	8	22	0	0	0	0
	5	N	0	0	29	0	15	0	0	8	15	8	0	29
			0.64	15	8	22	0	8	43	8	15	0	0	
		SE	0	0	8	22	0	8	43	8	15	0	0	0
	6	N	0	29	43	8	0	0	0	43	0	8	0	0
		SE	0.91	0	15	8	0	0	0	79	0	0	0	0
		SW	0.14	0	8	50	15	0	0	22	0	8	0	0
	7	N	0.29	0	60	0	14	0	0	0	14	0	0	14
		SE	1.33	27	40	27	0	0	0	34	0	0	0	0
		SW	0.9	0	34	14	7	7	7	40	0	0	0	0
	8	N	0.31	0	74	14	0	0	14	0	0	0	0	0
		SE	0.09	0	60	0	0	0	14	27	0	0	0	0
		SW	1	0	67	20	0	0	0	14	0	0	0	0
	9	N	0	0	43	22	0	0	0	22	8	0	0	8
		SE	0	0	43	22	15	0	0	0	15	8	0	0
		SW	0	0	50	15	0	0	22	15	0	0	0	0
	10	N	0	0	58	8	0	0	0	36	0	0	0	0
		SE	0	0	36	0	0	0	36	29	0	0	0	0
		SW	1.35	0	36	8	0	0	8	43	0	8	0	0

Table 1. Continued

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %	
Penhill	4	N	0.14	8	15	0	0	86	0	0	0	0	0	
		SE	0.25	50	8	8	0	86	0	0	0	0	0	
		SW	0.43	50	0	0	0	100	0	0	0	0	0	
	5	N	0.93	43	22	0	0	0	0	58	0	22	0	0
		SE	0.97	58	22	0	0	0	0	79	0	0	0	0
		SW	0.89	43	15	22	0	50	15	0	0	0	0	0
	6	N	0.45	15	15	22	8	58	0	0	0	0	0	0
		SW	0.5	29	29	15	0	36	15	0	8	0	0	0
			0.95	43	36	0	0	0	65	0	0	0	0	0
	7	N	0.06	15	15	43	0	29	0	0	0	15	0	0
		SE	0.22	29	50	8	0	36	0	0	0	8	0	0
		SW	0.5	58	43	22	0	36	0	0	0	0	0	0
	8	N	1.74	58	79	15	0	8	0	0	0	0	0	0
		SE	0.24	36	15	15	0	58	0	0	0	15	0	0
		SW	0.31	15	22	50	0	29	0	0	0	0	0	0
	9	N	1.37	29	22	0	0	15	65	0	0	0	0	0
		SE	2.41	36	15	15	0	8	65	0	0	0	0	0
		SW	0.85	0	0	58	0	22	15	0	8	0	0	0
10	N	1.62	15	72	0	0	15	0	0	0	0	15	0	
	SE	1.87	29	50	15	0	8	0	0	0	8	22	0	
	SW	1.66	36	50	15	0	8	0	0	0	8	22	0	

Table 1. Continued

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %	
Riverlands	4	N	0	0	34	0	40	14	0	0	0	14	0	
		SE	0	0	34	14	0	27	0	0	0	27	0	
		SW	0	0	54	0	20	7	0	0	0	0	20	0
	5	N	0	0	7	0	27	67	0	0	0	0	0	0
		SE	0	0	7	0	27	67	0	0	0	0	0	0
		SW	0	0	7	0	20	67	0	7	0	0	0	0
	6	N	0.11	0	54	0	0	0	0	0	0	0	47	0
		SE	0	0	74	0	0	0	0	0	0	0	27	0
		SW	0.09	0	54	0	0	7	0	0	0	0	34	7
	7	N	0	0	47	7	7	0	0	0	0	7	34	0
		SE	0	0	27	7	0	0	0	7	0	7	54	0
		SW	0	0	47	0	0	0	0	0	0	0	54	0
8	N	0	0	54	14	7	0	0	0	0	7	20	0	
	SE	0	0	40	14	0	7	0	0	0	7	34	0	
	9	SW	0	0	54	0	14	7	0	0	0	7	20	0
		N	0.13	0	40	14	0	14	0	0	0	0	34	0
		SE	0.13	0	27	14	7	0	7	0	0	0	47	0
	10	SW	0	0	47	7	0	0	0	0	0	0	47	0
		N	0	0	54	20	7	7	0	0	0	0	14	0
		SE	0	0	34	0	14	34	0	0	0	0	20	0
	11	SW	0	0	54	7	0	14	0	0	0	0	27	0
		N	0	0	7	54	14	14	0	0	0	0	14	0
			0	0	27	7	34	27	0	0	0	0	7	0
	0.09		0	34	40	7	14	0	0	0	0	7	0	
	12	N	0	0	47	0	7	14	0	0	0	7	27	0
		SE	0	0	14	14	0	54	0	0	0	0	20	0
SW		0	0	40	20	0	14	0	0	0	7	20	0	
13	N	0	0	54	7	0	0	0	0	0	0	40	0	
	SE	0	0	20	7	7	14	34	0	0	0	20	0	
	SW	0	0	40	7	0	0	0	27	0	14	14	0	

Table 1. Continued

Site	Point	Transect	Stem density (m2)	Canopy cover %	Bare ground %	Dead wood %	Herb/bulb %	Grass %	Leaf litter %	Perennial %	Shrub %	Restio %	Rock %	
Rooisands	1	N	1.65	47	0	0	0	67	27	0	7	0	0	
		SE	0.47	7	0	0	0	80	20	0	0	0	0	
		SW	0.08	0	34	0	0	0	60	0	0	7	0	0
	2	N	6.7	34	0	27	0	7	7	34	0	0	34	0
		SE	2.02	20	74	27	0	0	0	0	0	0	0	0
		SW	13.27	47	7	54	0	0	0	40	0	0	0	0
	3	N	6.72	50	50	36	0	15	15	0	0	0	0	0
		SE	0.68	22	43	8	0	29	29	0	0	15	8	0
		SW	2.83	43	29	15	0	22	22	0	0	8	29	0
	4	N	1.83	43	0	8	0	0	0	93	0	0	0	0
		SE	0.72	22	0	8	8	8	72	0	0	0	15	0
		SW	2.27	15	0	0	0	0	72	8	0	0	22	0
	5	N	1.14	50	36	15	0	15	15	0	0	8	29	0
		SE	0.85	22	43	0	8	8	0	0	0	0	50	0
		SW	1.7	22	72	0	0	0	15	0	0	8	8	0
	6	N	0	0	15	0	0	0	8	0	0	50	29	0
		SE	0.43	0	0	22	0	0	0	0	0	15	65	0
		SW	0	0	0	8	0	0	8	0	0	43	43	0
	7	N	1.22	79	0	0	0	0	29	72	0	0	0	0
		SE	3.22	65	0	0	0	0	43	58	0	0	0	0
		SW	2.33	43	0	22	0	0	58	22	0	0	0	0
	8	N	1.83	58	0	29	0	0	0	58	0	0	15	0
		SE	3.1	58	0	8	0	0	43	50	0	0	0	0
		SW	0.93	22	0	58	0	0	29	8	0	0	8	0
9	N	2.1	0	8	22	0	0	50	22	0	0	0	0	
	SE	1.45	0	29	8	0	0	36	22	0	8	0	0	
	SW	1.02	58	0	36	0	0	29	36	0	0	0	0	
10	N	1.58	36	0	43	0	0	0	0	0	0	58	0	
	SE	1.49	72	58	36	0	0	0	0	0	0	8	0	
	SW	5	54	7	27	27	27	0	40	0	0	0	0	

Appendix III. EVI data

Figure 1. Average EVI score per site per month for 2010.

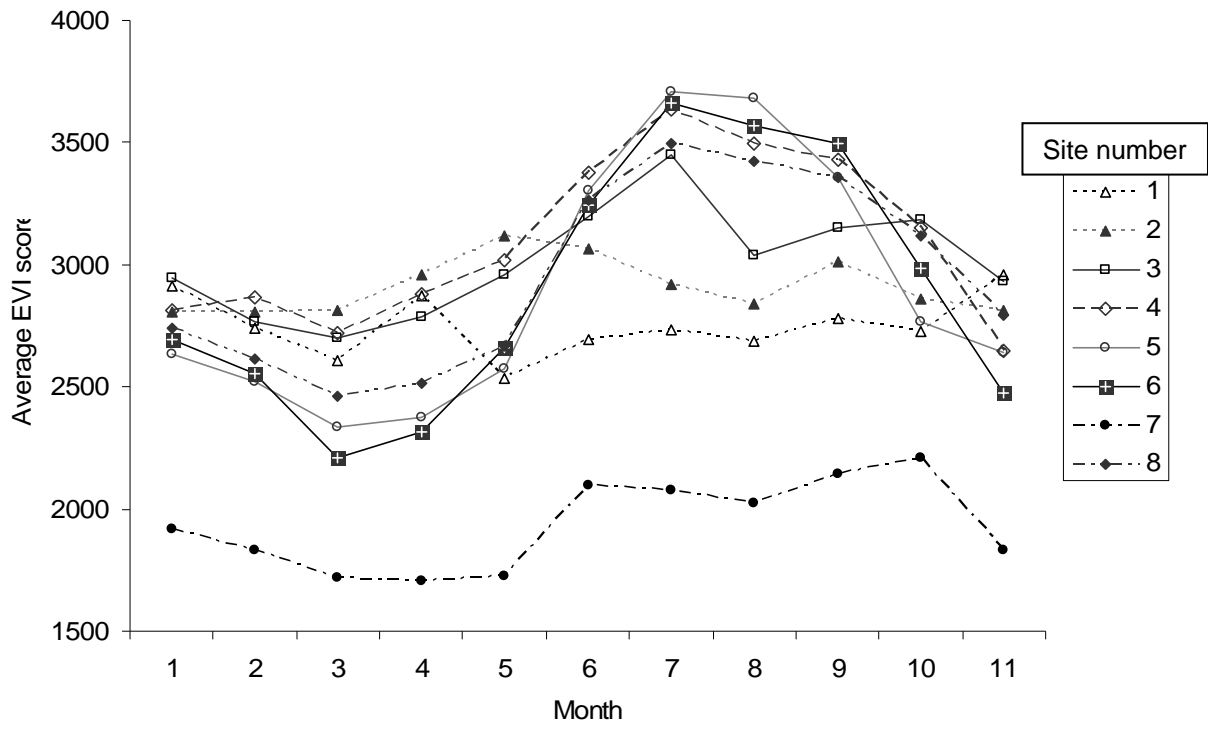


Table 1. EVI scores for each site organized by day and month. For more information see <https://edcsns17.cr.usgs.gov/NewEarthExplorer/>

Site	25 Jan	10 Feb	25 Feb	14 Mar	30 Mar	15 Apr	01 May	17 May	02 Jun	18 Jun	04 Jul	20 Jul	05 Aug	21 Aug	06 Sep	22 Sep	08 Oct	24 Oct	09 Nov	25 Nov
1	2501	2406	2886	3132	2373	3036	2905	2631	2813	2718	2713	2375	3176	2858	2295	2502	3056	3912	2558	2657
	2983	2609	2886	3472	2649	3454	2915	2631	2813	3112	2934	2783	3176	2858	2766	2808	3056	3912	3000	2950
	2703	2723	2827	3132	2456	3036	2663	2744	3148	2727	2636	2748	2444	2685	2766	3765	3056	3912	2435	3825
	2846	3081	3330	2853	3219	2985	2651	2913	3148	2806	3103	2739	2649	2685	2766	3533	2950	3158	3117	3597
	2522	2223	2553	2627	2225	3873	2651	2398	2800	2759	2795	2242	2428	2685	3312	3722	2420	2385	2588	3927
	2522	2370	2480	2627	2225	3873	2651	2528	2969	2607	2795	2499	2428	2685	3312	3767	2604	3158	2435	2743
	2913	2539	3040	2776	3564	3200	3567	2772	3115	2642	3833	3065	2795	4386	3946	3022	2782	3291	2761	4143
	2913	2539	2729	2520	3564	3200	2999	2948	3115	2707	3093	2980	2623	4386	3946	2766	3469	2872	2761	3927
	2653	2822	2729	2520	3564	3200	2773	2721	2800	2707	2929	2647	2655	3011	3946	3767	2854	2641	2951	3927
	2687	2872	2729	3443	2973	3820	2773	2562	2800	2523	2976	2499	2469	3011	2375	3767	2614	2574	2951	3927
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3281	2341	2988	3430	2882	2710	3584	2696	2881	2556	2939	2991	2988	2595	2415	2322	2923	3925	2652	2117
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1697	1721	2168	1801	1794	1633	1673	1770	2023	1856	1826	2012	1972	1870	1908	2264	2295	2068	1843	1872
1716	1721	1750	1693	1560	1614	1575	1790	2023	1984	1826	2012	1874	1833	2097	2064	2539	1935	1756	1751
1774	1651	1712	1693	1471	1802	1590	1697	2490	2306	2057	2164	1879	2247	2037	2325	2135	1940	2702	1616
1736	2352	1651	2021	1512	1897	2296	1806	2133	2221	2437	2474	2481	2247	2550	2634	2827	2125	2181	1688
2203	2208	1602	1871	1786	1592	1324	1745	2133	2253	2321	2501	2759	2426	2728	2718	2606	2088	1805	1701
1784	1665	1576	1503	1363	1436	1366	1678	2152	2005	1912	2221	2348	2282	2331	2353	2482	2025	1965	1757
2994	2218	2803	2584	2792	2607	2813	2644	3411	2592	2613	2656	2631	2359	2310	2715	3234	2678	2514	2018
2057	1972	3311	2262	2045	1856	1953	2193	2684	2136	2069	2258	2080	2034	2064	2131	2460	2068	2089	1946
2615	3310	3353	2619	2738	2674	2068	2369	3086	2423	3823	2258	2393	2037	3186	2467	3349	2453	2141	2206
2657	2694	1843	2344	2438	2729	2296	2644	3197	3094	2601	3551	3078	3008	3062	3343	2980	2623	2207	2242

	1791	1665	1576	1774	1312	1315	1441	1780	2339	2516	2110	2242	2401	2213	2846	2719	3231	2075	1964	1898
	3299	3398	3003	3247	2568	2774	2413	2704	2947	2382	2609	3645	2669	2250	2922	3109	3354	2023	3113	2949
8	2589	2860	2766	2840	2544	2405	2455	2942	3713	3550	3446	3138	3925	3384	3570	3394	3487	3286	3306	2619
	2741	2943	3035	2716	2625	2596	2657	2663	3713	3874	3926	4314	3696	3919	3925	2845	3222	2952	2949	2738
	2725	3074	2682	2557	2404	2557	2657	2942	3512	3799	3254	4403	3707	2906	3373	3657	3287	3418	3501	3192
	2460	2655	2287	2079	2065	2399	2657	2775	3333	3361	3558	3439	3630	3348	3343	3497	3413	2999	2552	2416
	2395	2657	2121	2535	2098	2328	2327	2318	3453	3209	3403	3439	3358	2679	2920	2857	3052	2942	2535	2426
	2742	2850	2203	2522	2343	2238	2454	2392	3052	2883	3403	2879	3210	2679	3283	3000	2763	2667	2482	2512
	2699	2673	2511	2355	2270	2622	2521	3100	3659	3695	4117	3726	3826	3390	3092	3843	3022	2961	2729	2677
	2448	2434	2232	2064	2100	2563	2521	2449	3453	3162	3558	3573	3506	3471	3357	3165	3181	2722	2729	2428
	2355	2568	2166	2064	2296	2333	2710	2679	3453	3155	3481	3599	3456	3509	3452	3128	3133	2766	2478	2479
	2610	2479	2478	2290	2338	2497	2675	2679	3325	3365	3481	3599	3483	3509	3350	3469	3133	2764	2293	2479
	2694	2442	2433	2415	2330	2409	2726	2457	3217	3512	3761	3359	3826	3349	3357	3565	3161	3264	3582	2755
	2448	2392	2342	2129	2026	2563	2726	2312	3295	2865	3340	3292	3240	3312	3357	3381	3228	2973	2733	2442
	2332	2512	2135	2262	2200	2294	2540	2607	3295	2895	3481	3542	3406	3449	3214	3250	3223	2799	2251	2496
	2822	2529	2913	2859	2424	2734	2638	2607	3325	3201	3145	3542	3387	3440	3190	3294	3223	2837	2701	2706
	2709	2862	2733	2570	2369	2663	2941	2822	3076	3395	3411	3424	3593	2842	3259	3200	3349	3148	2733	2656
	2196	2512	2101	2189	2104	2022	2222	2245	2833	2352	3150	2999	3157	3027	3214	3082	3105	2764	3077	2364
	2689	2579	2624	2432	2495	2611	3035	2817	3138	2879	3150	2999	3261	3092	2932	3289	3105	2962	2696	2695
	2814	2989	2793	3150	2619	3079	2758	2794	3428	3289	3731	3621	3588	3490	3420	3378	3170	2756	2875	2654
	2814	2887	3014	2831	2502	2811	2958	3024	3704	3042	3839	4021	3868	3527	3508	3605	3277	3205	3100	3005
	3006	3244	2877	2725	2892	3004	2907	3089	3336	3328	3839	3797	3613	3637	3665	3884	3695	3205	3417	2948
	2803	2574	2413	2367	2593	2161	2862	2457	2705	2368	2781	2967	3017	2986	3072	3191	3000	2819	3672	3044
	2473	2340	2560	2772	2262	2493	2517	2794	3428	3436	3220	3624	3729	3420	3590	3259	3457	3126	2949	2525
	2697	2560	2647	2632	2576	2498	2390	3024	3288	3735	4112	4021	3868	3646	3669	3309	3457	3040	3027	2672
	2896	2818	2915	2966	2980	3004	2694	2829	3407	3735	3010	2998	3895	3202	3739	3519	3428	3862	3234	3168