The effects of land use changes on the distribution of forest dependent bird species in South Africa

by
Tessa June Groves Cooper

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Supervisor: Prof. Michael I. Cherry
Co-supervisor: Andrew Wannenburgh
Faculty of Science

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Declaration

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Abstract

Forests in South Africa have had a long history of human utilization and disturbance, and are under threat from a variety of anthropogenic land use changes. Foremost of these are deforestation and forest degradation, impacting the species native to these forests. The aims of this study were to determine changes in the distribution of forest dependent bird species according to the South African Bird Atlas Project; to relate these changes to changes in land-use; to identify links between these changes; to determine the extent, location and causes of the decline of each forest dependent bird species; and to identify current risks to forest dependent bird species in South Africa. Range data on 57 forest dependent bird species from SABAP1 (1987-1992) and SABAP2 (2007-present) were analyzed. Of these, 28 species were found to have declining ranges. Thirty sites across South Africa were identified as being most at risk, with all having experienced a loss of more than 10 of the 57 forest dependent bird species between SABAP1 and SABAP2. The range change data of the 28 species with decreasing ranges were correlated with data on changes in land cover over the same time period to infer relationships between changes in land use and change in bird ranges. Occupancy modelling was done to determine which land cover types affect extinction and initial presence. Individual species characteristics were analyzed to determine links between characteristics and response to land use change. A pan-European trait-based risk assessment framework was applied to all 57 species to identify habitats and species most at risk, as well as the most important threats to species persistence. Results showed that natural vegetation decreased in 67% of sites, while plantations and cultivation increased in 50% of sites. Occupancy modelling showed extinction likelihood to increase with plantations in some species, while plantations mitigated extinction likelihood in other species. Urbanization and cultivation likewise mitigated extinction likelihood in some species. Natural vegetation was replaced by cultivation, while cultivation was replaced by urbanization. The number of species lost increased with a loss of natural vegetation. Twenty two of the thirty sites experienced deforestation of indigenous forests between 2000 and 2013/2014; changes in natural vegetation in these sites can be attributed primarily to deforestation, and a loss of plantations. While most at-risk sites were in the Eastern Cape, there was no geographic grouping of

species loss or of land use change. Most species lost were birds of prey or insectivores, and species characteristics and habitat preferences determined the sites from which they were lost. The Cape parrot (*Poicephalus robustus*), rufous-chested sparrowhawk (*Accipiter rufiventris*) and the migratory Eurasian golden oriole (*Oriolus oriolus*) suffered the largest declines in range size and are thought to be most at risk. Montane forests were found to be more at risk than other forest types. The major risks facing montane forests were increased abundance of small predators, increased fire suppression, increased soil management, removal of deadwood and reduced diversity of tree species. These threats are all products of plantation forestry and local harvesting. Nesting risk was higher than foraging risk for all species, indicating that nesting habitat should be better preserved. Half of South Africa's forest dependent bird species have declining ranges, with the loss of these species most prominent in the Eastern Cape. Natural vegetation loss, comprising mostly recent deforestation; increased cultivation and urbanization; and changes in plantation cover are thought to be the main factors determining these declines. Montane forests in particular should be better protected to preserve forest dependent species, and the negative effects of plantation forestry and local harvesting should be mitigated.

Opsomming

Suid-Afrika het 'n lang geskiedenis van woude wat deur mense gebruik, asook vernietig word. Woude word bedreig deur verskeie menslike grondgebruike en veranderinge. Die mees vooraanstaande van die bedreigings is ontbossing en woud verdunning, wat 'n invloed het op inheemse spesies in die woude. Die doelwitte van hierdie studie was om veranderinge in die verspreiding van bos-afhanklike voëlspesies vas te stel volgens die Suid Afrikaanse Voël Atlas Projek; om hierdie veranderinge te verband met veranderinge in grondgebruik; om verbande tussen hierdie veranderinge te identifiseer; om die mate, die plek en die oorsake van die agteruitgang van elke bos-afhanklike voëlspesies te bepaal; en om huidige risiko's aan die bos afhanklik voëlspesies in Suid-Afrika te identifiseer. Trefwydte inligting oor 57 woud afhanklike voëlspesies van SABAP1 (1987-1992) en SABAP2 (2007-nou) was geanaliseer. Daar was gevind dat vanuit die spesies, het 28 'n afneming in trefwydte ervaar. Dertig terreine in Suid-Afrika was uitgeken as dié met die hoogste risiko, met 'n verlies van meer as 10 van die 57 woud afhanklike voëlspesies tussen SABAP1 en SABAP2. Die trefwydte inligting van die 28 spesies met 'n afnemende trefwydte was gekorreleer met inligting oor veranderinge in grond bedekking oor dieselfde tydperk om verhoudings tussen veranderinge in grond gebruik en veranderinge in voël trefwydtes aftelei. Besetting modellering was gedoen om te bepaal watter tipes grond bedekking beinvloed uitwissing en aanvanklike teenwoordigheid. Kenmerke van individuele spesies is ontleed om verbande tussen kenmerke en reaksie op verandering in grondgebruik te bepaal. 'n Pan-Europese eienskap gebaseerde risiko-analise raamwerk is toegepas op die 57 spesies om die spesies en habitatte met die grootse risiko te identifiseer, asook die belangrikste bedreigings vir spesies-volharding. Die resultate het gewys dat natuurlike plantegroei het verminder in 67% van terreine, terwyl plantasie en verbouing vermeerder het in 50% van terreine. Besetting-modellering het gewys dat waarskynlikheid van uitwissing vermeerder met plantasies in sommige spesies, terwyl plantasies die waarskynlikheid van uitwissing verminder het in ander spesies. Verstedeliking en verbouing het ook die waarskynlikheid van uitwissing verminder in sommige spesies. Natuurlike plantegroei was vervang deur verbouing, terwyl verbouing vervang is deur verstedeliking. Die aantal spesies verlies het vermeerder met die vermindering van natuurlike

plantegroei. Twee en twintig van die dertig terreine het ontbossing van inheemse woude ervaar tussen 2000 en 2013/2014. Veranderinge in natuurlike plantegroei in die terreine is meestal as gevolg van ontbossing, en 'n vermindering van plantasies. Terwyl meeste van die hoë risiko terreine in die Oos-Kaap was, was daar geen geografiese groepering van spesies vermindering of grondgebruik veranderinge nie. Meeste spesies wat vernietig is was roofvoëls of insectivore, en spesies kenmerke en habitat voorkeure het die terreine bepaal waaruit hulle verloor is. Die grootpapegaai (*Poicephalus robustus*), rooiborssperwer (*Accipiter rufiventris*) en die trekvoël Europese wielewaal (*Oriolus oriolus*) het die grootste vermindering in trefwydte grootte ervaar, en is vermoedelik die grootste risiko. Bergwoude was gevind om meer in gevaar te wees as ander woud tipes. Die grootste risikos wat bergwoude beïnvloed het was 'n toenemende hoeveelheid klein roofdiere, verhoogde vuur onderdrukking, verhoogde grondbestuur, verwydering van dooie hout, en verlaagde diversiteit van boomspesies. Hierdie bedreigings is almal as gevolg van plantasie bosbou en plaaslike oes. Nes-risiko was hoër as kos soek risiko vir alle spesies, wat aandui dat nes-habitat beter bewaar moet word. Die helfte van Suid-Afrika se woud-afhanklike-voëlspesies het dalende trefwydtes, met die verlies van hierdie spesies mees opvallend in die Oos-Kaap. Die verlies van natuurlike plantegroei, as gevolg van onlangse ontbossing; verhoogde verbouing en verstedeliking; en veranderinge in plantasie bedekking is vermoedelik die hooffaktore wat die vermindering van voëlspesies veroorsaak. Veral bergwoude moet beter beskerm word om woud-afhanklike-voëlspesies te bewaar en die negatiewe invloede van plantasiebosbou en plaaslike oes te verminder.

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Chapter 1 - Introduction

Forests and forest biodiversity globally

Forest ecosystems cover approximately 30% of the Earth's terrestrial surface (FAO and JRC 2012), and contain some of the highest biodiversity of all biomes, with tropical forests alone containing more than 60% of terrestrial biodiversity (Gardner *et al.* 2009). However, these ecosystems are particularly threatened by human activities, with more than 50% of forests globally being compromised (FAO 2009; Chama *et al.* 2013a). The FAO and JRC (2012) estimated that, between 1995 and 2005, all major forest biomes globally had suffered a significant decline in land mass covered. The biggest anthropogenic threats to forests worldwide include deforestation, forest degradation, fragmentation, over-harvesting, altered fire regimes and invasion by alien plants and animals (Gardner *et al.* 2009; Tabarelli *et al.* 2004; Atkinson 1977). Human-induced climate change is also a major threat (Gardner *et al.* 2009). The impacts of these changes differ depending on the characteristics of the remaining intact forest patches (Chama *et al.* 2013a), as well as on the characteristics of the species within them (Farwig *et al.* 2008). A loss of certain species within a forest patch could have consequences as diverse as trophic level cascades, limitation of functional processes through the interruption of species connecting habitats functionally, and facilitation of invasion (Gardner *et al.* 2009). The effects of these threats on forest birds globally include increased predation (Atkinson 1977), increased brood parasitism (Robinson *et al.* 1995), population declines in migrant species (Robinson *et al.* 1995), and decreased reproductive success (Paton 1994). Forest patch size and structure have been found to influence bird species abundance and persistence (Robbins *et al.* 1989), as have landscape structure, forest cover and forest configuration, with these effects often being species-specific (Villard *et al.* 1999).

African forests have suffered some of the highest levels of deforestation globally, with a mean annual loss of 1.1 million ha between 1990 and 2000, and 2.7 million ha between 2000 and 2005 (FAO and JRC 2012). A study by Barnes and Lahm (1997) found tropical African forests to be less productive than forests globally, with a low rate of production of edible plant biomass and consequently a lower

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biomass of herbivores than other local ecosystems. Threats to African forests include subsistence hunting, shifting cultivation (Barnes and Lahm 1997), and fragmentation. The latter has led to declines in birds and animal-dispersed tree species (Cordeiro and Howe 2003, 2001); a loss of rare, forest interior and forest understory birds (Newmark 1991), in Tanzania. Patch size, canopy structure and abundance of large trees have been shown to affect bird species richness in Ghana (Beier *et al.* 2002).

South African forests

Indigenous forests in South Africa are estimated to comprise approximately 7177km^2 (~0.56%) of the country's landmass (Low and Rebelo 1996). This biome is naturally highly fragmented (Low and Rebelo 1996), and forms an archipelago of land islands (Lawes *et al.* 2004a), the largest of which is the Knysna-Tsitsikamma forest in the Western Cape (Geldenhuys 1991). The patchy nature of the forest biome is largely attributed to palaeoclimatic changes (Lawes *et al.* 2004a; Berliner 2009), and is thought to reflect the natural limits of the biome (Kruger *et al.* 1995; McCracken 2004; Willis 2004).

Delineation of forest subtypes in South Africa has undergone many reclassifications, with the earliest based on timber quality and quantity (von Maltitz *et al.* 2003). Generally classifications split forest types into two larger categories; that of the inland Afromontane or Afrotemperate forests and the coastal Indian Ocean types (Lawes *et al.* 2004a). Subdivision by Cooper (1985) split forest into ten types; Acocks (1953) split forest into 15 types and subtypes; von Maltitz *et al.* (2003) split forest into 24 types, while Mucina and Rutherford (2006) used twelve broader categorizations. These can be seen in Figures 1.1-1.3 on the following pages.

Vegetation Types (Mucina & Rutherford, 2006)
LEGEND
FOa 1 Lowveld Riverine Forest
FOa 2 Swamp Forest
FOa 3 Mangrove Forest
FOz 1 Southern Afrotemperate Forest
FOz 2 Northern Afrotemperate Forest
FOz 3 Southern Mistbelt Forest
FOz 4 Northern Mistbelt Forest
FOz 5 Scarp Forest
FOz 6 Southern Coastal Forest
FOz 7 Northern Coastal Forest
FOz 8 Sand Forest
FOz 9 Ironwood Dry Forest

Figure 1.1 The forest types of the Western Cape province of South Africa, after Mucina and Rutherford (2006).

Vegetation Types (Mucina & Rutherford, 2006)
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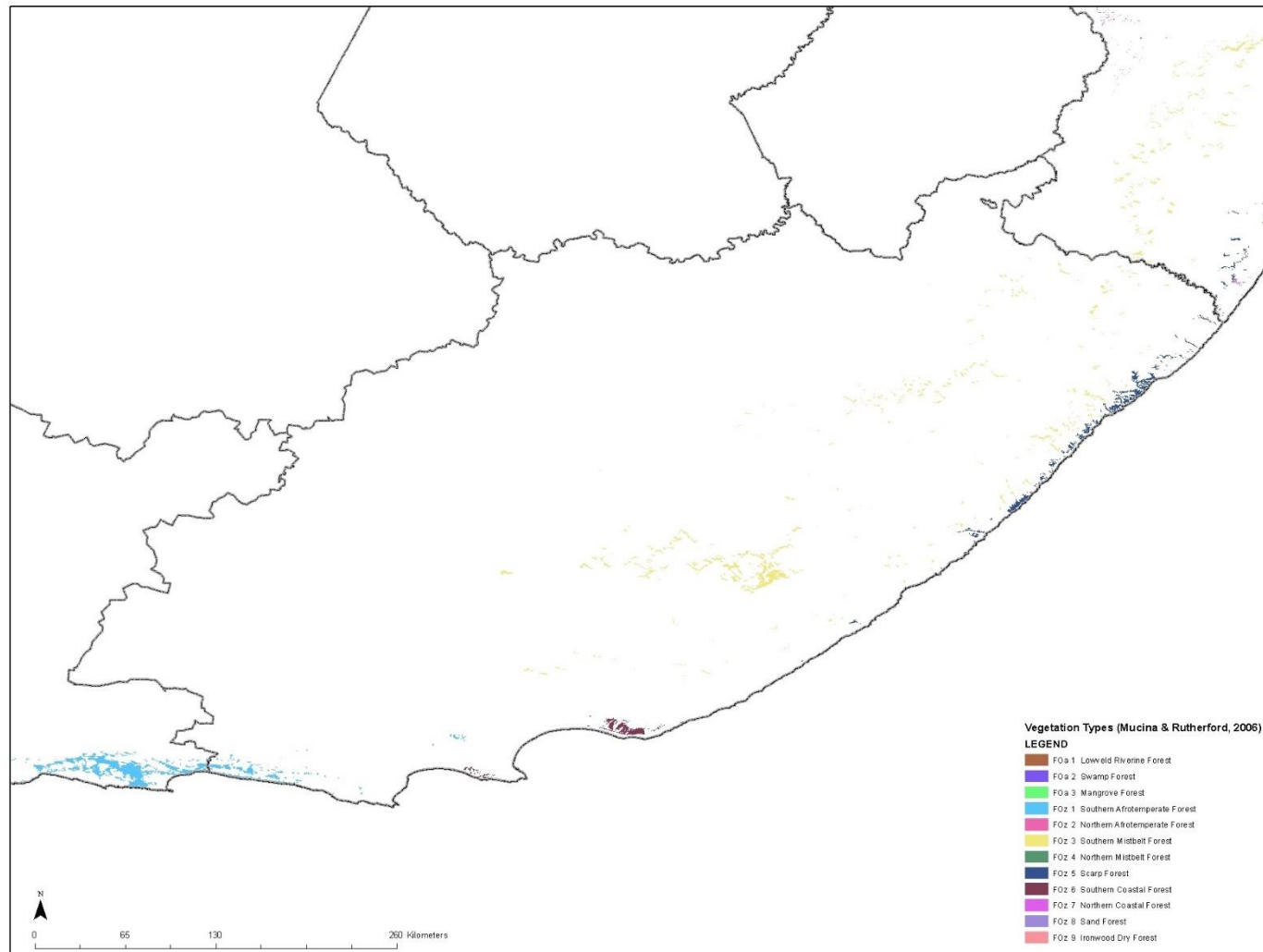
Figure 1.2 The forest types of the Eastern Cape province of South Africa, after Mucina and Rutherford (2006).

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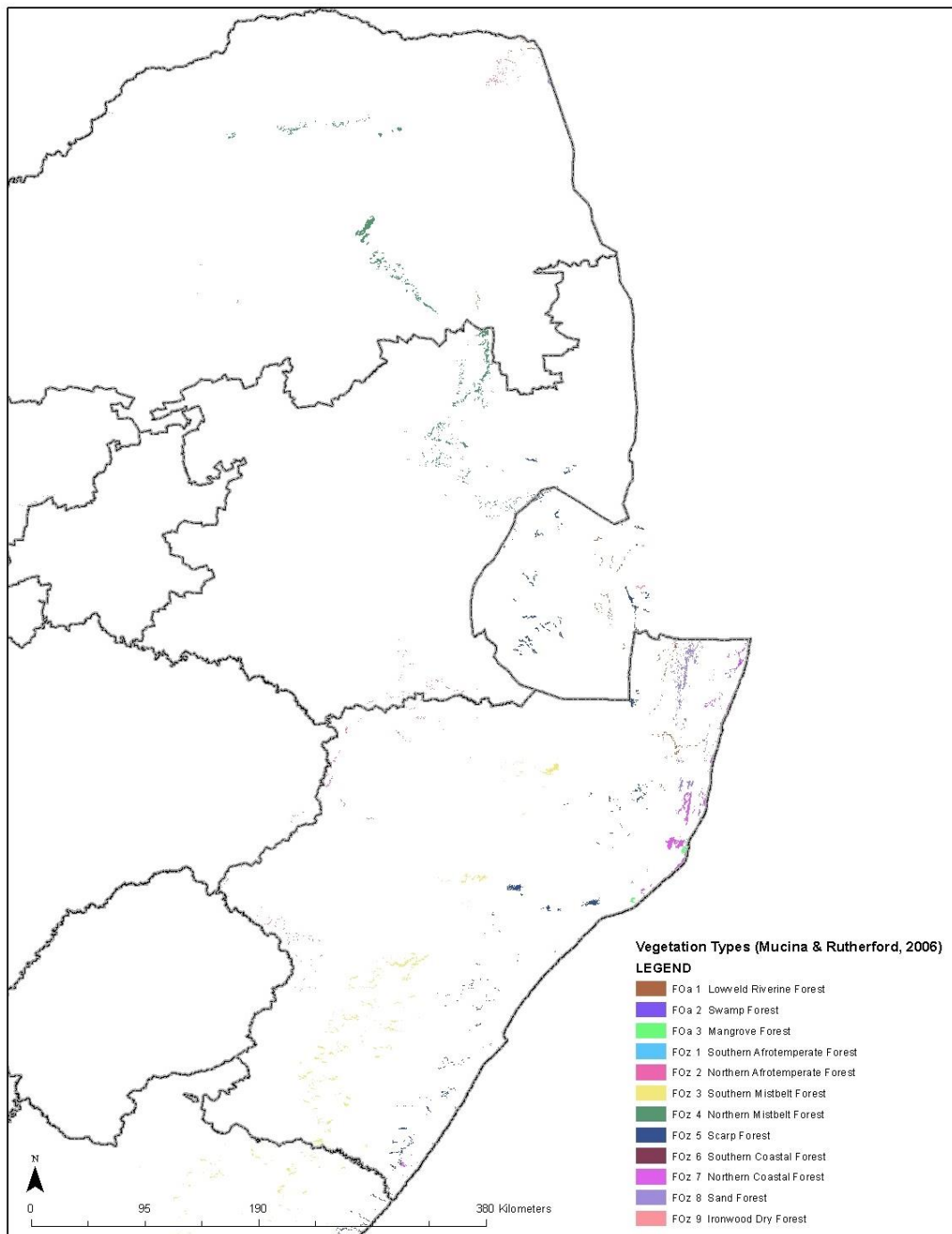
Figure 1.3 The forest types of the KwaZulu-Natal, Mpumalanga and Limpopo provinces of South Africa, after Mucina and Rutherford (2006).

South African forests have high biodiversity, with the highest plant species density per unit area, and the second highest tree species richness of temperate forests globally (Berliner 2009). The number of threatened species in South African forests is likewise high, with forests supporting the largest percentage per 1000 km^2 of threatened bird, mammal, reptile and amphibian species of any biome in South Africa (Berliner 2009). The intactness of this biodiversity is essential to ecosystem functioning (Berliner 2009). The fragmented nature of South African forests, as well as the high propensity for zoochory (seed dispersal mediated by animals (Lawrence 2008)) found in the tree species present, mean that a positive feedback loop is likely between increasingly fragmented forest patches and declines in bird numbers: birds may enhance connectivity between forest patches by facilitating seed dispersal, but increased fragmentation of the forests makes it more difficult for faunal species to traverse the inter-forest matrix, which furthers fragmentation by limiting tree dispersal (Berliner 2009; Trimble and van Aarde 2011).

Forest protection and utilization in South Africa

Forests have been protected in South Africa from as early as 1883 (Willis 2004), with increased protection in the last three decades (Donald and Theron 1983; Downs and Symes 2004). The beginning of the apartheid era led to the delegation of forest protection in each former 'homeland' to their relevant governments, while the remainder of the country's forests were under the protection of the National Forestry Department (von Maltitz and Shackleton 2004). After 1994, forests in South Africa were managed either by the state, private organizations or communally, by the relevant tribal authority (von Maltitz and Shackleton 2004). These tribal authorities were traditionally the headmen, who controlled the use of forest resources in pre-colonial times. However, this role was weakened during colonial rule and is consequently less effective at present (von Maltitz and Shackleton 2004). Currently forests are protected by the National Forests Act (1998, Act 84) and the Forestry Laws Amendment Act (2005, Act 35).

Indigenous forest in South Africa has been utilised by humans since early colonization of the Cape in the 17th century, although prior utilisation by native people is likely to have occurred (Feely 2004;

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Willis 2004). This use intensified through the 18th and 19th centuries, until the first forest regulation limiting use and degradation was put in place in 1883 (Willis 2004). To protect the indigenous forests from unsustainable use and supply the growing timber demand, exotic plantations were introduced in the late 1800's (von Maltitz and Shackleton 2004; Scholes *et al.* 1995). Aside from timber use by settlements, forests were utilized by the military for wood products and fuel; cleared and utilized for roads and communication systems; and fuelwood and charcoal were utilized for the mining industry (McCracken 2004).

Forest resources are important for both their utilization (e.g. timber, fuelwood and other non-timber products) and non-use value (e.g. recreation and tourism, cultural and spiritual) (Chipeta and Kowero 2004). Non-timber forest products include plants used for traditional medicine, gathered foods, including honey and hunting, handicrafts and household items, and fuelwood, with more than 50% of domestic energy in South Africa produced from fuelwood (Lawes *et al.* 2004b). In addition, forests provide ecosystem services such as water retention, soil conservation, and genetic resources (Cunningham *et al.* 1988). The harvesting of non-timber forest products is thought to be one of the largest threats to forests globally (Cocks *et al.* 2004), and plant use for medicine (Cocks *et al.* 2004) and animal use for food (White 2004) form a large part of this use.

Fire has long had an impact on indigenous forest in South Africa, with evidence suggesting that the current distribution of forests in South Africa is due to fire (Geldenhuys 1994). These fires were both natural and caused by humans (McCracken 2004), with anthropogenic fires threatening natural forest distributions. More recently, threats to forests include unsustainable use and exploitation of forest resources by rural peoples (Willis 2004). Local collection and harvesting is common in South Africa's indigenous forests, and is an important part of rural livelihood (Cocks and Wiersum 2003; Shackleton and Shackleton 2004). However, this harvesting is unsustainable, particularly in the Eastern Cape, where the amount of deadwood removed for use as fuelwood annually approximates the amount produced by the indigenous forests (Obiri 2002). In addition, rural farmers winter their cattle in forests (McCracken 2004), and this in combination with harvesting of fuelwood and plant material

(Cocks and Wiersum 2003; Shackleton and Shackleton 2004) leads to the degradation of the forest understory. In the past, forests were cleared for agriculture, settlements and plantations (Feely 2004; Berliner 2009), leading to anthropogenic fragmentation. Indigenous forests still suffer the threat of anthropogenic fragmentation (Lawes *et al.* 2006; Berliner 2009), thought to be amplified by the abundance of plantations in the matrix surrounding indigenous forests (Berliner 2009).

Plantation forests in South Africa

Plantations cover twice the area that indigenous forests do: 1 268 443 ha, or 1.1%, of South Africa's surface area (Forestry Economics Services CC 2014). Originally planted to benefit indigenous forests by providing an alternative source of timber (von Maltitz and Shackleton 2004), whether their utility outweighs their biological consequences is now less clear. The benefits of plantations to indigenous forests include improving biodiversity and reducing extinction in small forest patches (Wethered and Lawes 2003, 2005); acting as sheltering nurseries for saplings of indigenous trees (Geldenhuys 1997); and provision of a refuge for certain species when their natural habitat is denuded and fragmented, such as birds of prey, which are vulnerable to fragmentation because of their large body size (Ewers and Didham 2006) and are often found in plantations (Prestt 1965).

The negative impacts of plantations include causing forest degradation, leading to a reduction in biodiversity of adjacent forest patches (Geldenhuys 1991; Wethered and Lawes 2003, 2005; Najera and Simonetti 2009); the alteration of species assemblages (Armstrong and van Hensburgen 1995; Allan *et al.* 1997; Estades and Temple 1999); and the alteration of vegetation structure due to fire suppression, affecting species dependent on forest understory vegetation (Esseen *et al.* 1997; Brawn et al. 2001; Berliner 2009; Betts *et al.* 2010). Alien invasion has been shown to be facilitated by plantation forestry (Geldenhuys 1997; Richardson 1998; Van Wilgen *et al.* 2001); such invasion is thought to be one of the main causes of the thinning of indigenous forest cover (DAFF 2011).

Impacts of land use change on biodiversity

The response of individual species to habitat loss and fragmentation may differ by species characteristics. Factors influencing a species' response to habitat fragmentation and loss are trophic level, niche breadth, rarity (Ewers and Didham 2006), dispersal ability (Ewers and Didham 2006; Hinsley *et al.* 2009), degree of specialization (Hinsley *et al.* 2009), range size and body size (Trimble and Van Aarde 2011). Response to forest modification and disturbance also differs with the level of forest specialization displayed by the species (Neuschulz *et al.* 2011). The ability of species to move across a human-modified matrix is dependent on both functional guild specificity and the nature of the landscape matrix (Neuschulz *et al.* 2012).

Land use conversion to agriculture and timber plantations has occurred to a large scale in South Africa historically (Berliner 2009), and such changes in land use are one of the major factors determining species distributions. Certain species have adapted to these changes, for example by utilizing suburban gardens as alternative habitats. A recent study by Fairbanks *et al.* (2004) found species richness to be higher in human-modified landscapes where remnants of original habitats may be found, with both specialists of the original habitat and those that could adapt to the transformed landscape present.

South African forest birds

Some of the first work on South African bird species was done in the Western Cape, Eastern Cape and KwaZulu-Natal provinces. Cooper (1985) completed transects of the KwaZulu-Natal forests, and compiled a list of forest birds found here, as well as the forest types they occur in. Oatley (1989) examined the assumptions surrounding forest birds, and created a list of forest dependent birds, used as a baseline in this study. Cody (1983) examined the bird diversity and density of Eastern Cape and Western Cape forests, and found decreased species diversity in the forest patches of the Western Cape. Koen (1988) developed a new census technique to examine the bird communities of Afromontane forest, using the Knysna forest in the Western Cape as a study site. Armstrong and van Hensbergen (1995) found the bird diversity in Grootvadersbosch, Western Cape, was lessened by the

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presence of plantations and the action of clearfelling. Brown (2006) studied the bird species richness of a forest in KwaZulu-Natal between 1996 and 2003, and found that species richness increased in summer with seasonal migrants, while forest specialists were most abundant in winter due to the influx of regional and altitudinal migrants, in contrast to the findings of other studies on the changes in richness of forest-specific species throughout the year (e.g. Symes *et al.* 2002). Downs and Symes (2004) likewise examined the forests of KwaZulu-Natal to determine the necessity of snags to the reproduction of cavity-nesting bird species. Grass *et al.* (2013) found frugivorous bird response to habitat loss and alien invasion in KwaZulu-Natal to differ with species guild. This was especially true for forest specialists in sites with increased habitat loss. Lenz *et al.* (2015) provided the first work on the processes involved in the seasonal range expansion or nomadism of frugivorous birds in a study in KwaZulu-Natal. The avifauna of the forests of the Northern Province was studied by Symes *et al.* (2000), who found commercial afforestation to afford a buffering effect to patches of indigenous forests. Individual species studied have included the red-breasted sparrowhawk *Accipiter rufiventris* (Simmons 1986), the orange ground-thrush *Zoothera gurneyi* (Earlé and Oatley 1983), and extensive work on the Cape parrot *Poicephalus robustus* (Downs 2005; Downs *et al.* 2014; Wirminghaus *et al.* 1999; 2000; 2001a; 2001b; 2002).

The South African Bird Atlas Project

The first South African Bird Atlas Project (SABAP) was run from 1987-1992, and collected presence/absence data on bird species for each 15' by 15' quarter-degree grid cell of South Africa, Namibia, Botswana, Zimbabwe, Lesotho and Swaziland. The surveying technique was to list all the bird species seen in a quarter-degree grid cell within 30 days. The second SABAP was run from 2007-present, although in this study data up until September 2014 were used. It included South Africa, Lesotho and Swaziland, and later Namibia. Presence/absence data were collected per 5' X 5' pentad, nine of which make up a quarter-degree grid cell used in SABAP1. The surveying technique was to list all the bird species recorded in a pentad within five days. The two SABAPs can be compared if the results from the nine pentads making up a quarter degree grid cell are combined. The finer scale of sampling in SABAP2 leads to the possibility of inferring false increases in range. This is because only

a small portion of the larger QDGC used in SABAP1 would need to be visited and surveyed in order for the QDGC to be marked as surveyed, with a potential bias towards easy-to-access locations. In SABAP2, however, the much smaller pentads surveyed make it more likely that the majority of a QDGC, including those less than easy-to-access locations, would be surveyed. Thus it is more possible that a species was missed in SABAP1 (due to residing in impenetrable thicket in a remote corner of the QDGC, for example) and consequently surveyed and marked as "increased in range" in SABAP2, than the converse happening. Thus estimates of decreases are likely to be conservative for the same reason. Additionally, any vagrant or unusual species were assessed by a team of regional experts in both SABAP1 and SABAP2 before inclusion in the records for the particular QDGC was considered (Harebottle *et al.* 2010). When overlaid with maps of changing forest distribution and changes in land use over the same time period, reasons for declines in distribution may be inferred.

The abundance and availability of data from the first and second SABAP have allowed a variety of studies, on topics from indicator species (Reyers *et al.* 2000; Bonn *et al.* 2002), conservation priority areas (Bonn and Gaston 2005), Karoo bird diversity (Dean 1997, 2000), land use changes (Fairbanks *et al.* 2002; Fairbanks 2004), reserve selection (Fairbanks *et al.* 2001; Reyers et al. 2002; Gaston and Rodrigues 2003), climate change (Walther and van Niekerk 2015), distributions and dispersal (Péron and Altwegg 2015b), as well as individual species studies, such as that of Hofmeyr *et al.* (2014) on the secretarybird *Sagittarius serpentarius*. Other impacts of the SABAP data have been to allow designation of Important Bird Areas (IBAs) (Barnes 1998), and update Red List data (Barnes 2000) and species distribution data (Harrison *et al.* 1997).

Allan *et al.* (1997) used the first SABAP data to determine the impacts of plantations on grassland bird populations in Mpumalanga, and found that the addition of plantations to an area changes the assemblage of birds. Specifically, afforestation of grasslands with plantations allows species more common in wooded habitats and forests to colonise these areas. Péron and Altwegg (2015a) examined the changes in distribution of passerines and found that bush encroachment led to changes in species assemblages, as did agricultural conversion. Loftie-Eaton (2014) likewise found that species

inhabiting wooded habitats had the greatest observed increases between the two SABAP periods, while habitat specialists and species vulnerable to anthropogenic disturbance had the smallest observed increases.

Research rationale

Forest dependent bird species in South Africa are under threat, with the number of species listed as near-threatened or above on the IUCN Red List increasing from 10% to 19% between 2009 and 2014 (Berliner 2009; BirdLife South Africa 2014). Habitat loss caused by deforestation and fragmentation of indigenous forests are the largest threats to these species (Berliner 2009), and put the 57 forest dependent birds studied here at risk from habitat loss. It is vital to these species' conservation to understand how landscape-level changes and individual risks affect them, to mitigate further species declines.

This research will improve the understanding of links between changes in land use and bird ranges; provide a review of forest dependent bird species and risks to their persistence; and provide insight into South Africa's indigenous and plantation forests. The results of this study will inform management programmes and give an indication of the appropriateness of current forestry practices in South Africa, as well as possible areas for improvement. The determination of range changes of threatened species can give an indication of how well current conservation initiatives are working. Site identification will suggest forest fragments which are in need of better protection, allowing a greater understanding of where conservation funding and effort should be concentrated. Species-specific risks and characteristics studies should identify which species are less able to adapt to habitat disturbances, and which disturbances put them most at risk, allowing further understanding of conservation needs.

Aims of study

Along with recent changes in forest distribution due to deforestation of indigenous forests and afforestation with plantation forests in South Africa, many bird species dependent on indigenous

forests or forest fragments may have experienced range shifts. This is demonstrated in the differences in species ranges mapped by SABAP 1 and 2. It is posited that changes in forest distribution have caused changes in the distributions of forest-dependent bird species. Initial overview of the SABAP data show range retraction in some species, range shifts in others, and range expansion in others. Understanding the mechanisms underlying these changes is important in the conservation of both threatened and habitat-specific bird species, as well as of remaining indigenous forest patches.

The research aims and objectives of this project were as follows:

1. To determine declines in the distribution of forest-dependent bird species in South Africa during the time period of twenty years between the two SABAP surveys.

2. To relate these changes to changes in land-use, specifically deforestation of indigenous forest and afforestation with alien plantations, which occurred over the same time period.

3. To identify specific risks threatening forest bird species in South Africa.

4. To identify patterns in the characteristics of declining forest birds, and patterns in the location of forest bird declines, which may give insight into future trends.

The assumptions of this study are that the SABAP presence-absence data give a reasonably accurate depiction of changes in bird distribution over the last twenty years, and that the data acquired from the various national land cover datasets used are reasonably accurate. It is fair to assume that some error lies in both these datasets; however, no alternative datasets for such a large group of species or land cover types, for the entirety of the country, exist to the best of the author's knowledge.

Chapter overview

The data chapters are presented as manuscripts as follows:

Chapter 2 – *The effects of land use change on forest dependent bird declines in South Africa*

The 57 forest dependent bird species of South Africa were identified, as were the trends in their ranges using SABAP data. The 28 with decreasing ranges were used for further analyses in this

chapter. Thirty sites across South Africa were identified as at-risk, as more than ten forest dependent bird species were lost from them in the period between SABAP1 and SABAP2. The changes in land cover occurring in these sites over the same time period was determined, and this was used to model occupancy to determine which changes in land cover affect each species.

Chapter 3 – *A species assessment of declining South African forest dependent birds*

Each of the 28 species with decreasing ranges were examined to determine which specific land use changes have been affecting them, where they have been declining, and postulates reasons for these declines. Patterns in the suite of species lost and the geographic location of losses were identified here.

Chapter 4 – *A trait-based risk assessment of South African forest birds*

A pan-European risk assessment framework was applied to all forest dependent South African bird species to determine the major risks facing these species, as well as the species most at risk The original framework was furthered through the addition of a 'net risk' model, which incorporates any benefits of land use change on species into the original model.

The data chapters are followed by a concluding chapter. This chapter synthesizes the results of chapters 2-4, and makes recommendations for future conservation measures.

Chapter 2 - The effects of land use change on forest dependent bird declines across South Africa

Cooper, TJG[1]; Wannenburgh, A[2]; Cherry, MI[1]

[1]Department of Botany and Zoology, Stellenbosch University; [2]Department of Environmental Affairs, South Africa

Abstract

Forest ecosystems in South Africa are at risk from a variety of anthropogenic threats, impacting the faunal species dependent on them. Range data on 57 forest dependent bird species from SABAP1 and SABAP2 were analysed. Of these, 28 were found to have declining ranges, and these data were correlated with data on changes in land use over the same time period acquired by comparing land cover data from 1994 and 2012. Occupancy modelling to determine which land cover types affect extinction was done for 30 sites across South Africa which experienced a loss of more than 10 species. The data examined showed the Cape parrot (*Poicephalus robustus*) to have suffered the largest loss in range (-58%). Natural vegetation decreased in 67% of sites, while plantations and cultivation increased in 50% of sites. Occupancy modelling showed extinction to be both positively and negatively related to plantations and negatively related to urbanization and cultivation. Natural vegetation was replaced by cultivation, while cultivation was replaced by urbanization. The number of species lost increased with a loss of natural vegetation. Responses to land use change did not appear to be related to particular species characteristics. Twenty two of the thirty sites experienced deforestation of indigenous forests between 2000 and 2013/2014. Changes in natural vegetation in these sites can be attributed primarily to deforestation, while for the remaining eight sites changes in the extent of both forest and plantations occurred between 1994 and 2000. Half of South Africa's forest dependent bird species have declining ranges, with the loss of these species most prominent in the Eastern Cape province. Natural vegetation loss, comprising mostly recent deforestation; increased cultivation and urbanization; and changes in plantation cover are thought to be the main factors determining these declines.

Introduction

Forest habitats make up approximately 0.56% of South Africa's landscape, but are home to some 14% of the country's terrestrial bird species, many of which are endemic, and seven of which are range-restricted endemics found only in these forests (Geldenhuys and MacDevette 1989; Low and Rebelo 1996; BirdLife International 2013). Natural fragmentation occurred as a consequence of climate changes during the Quaternary which resulted in contractions and expansion of forests, so their biota has evolved under these conditions (Eeley *et al.* 1999; Kotze and Lawes 2007). Recently forest habitats have been extensively further fragmented by human activities, with most remaining forests being smaller than 1 km^2 (Eeley *et al*. 1999). Fragmentation is largely the result of deforestation, both for commercial plantations and by rural communities, with nearly 50% of indigenous forests in South Africa estimated to be degraded as a result of anthropogenic fragmentation (MacDonald 1989; Eeley *et al*. 1999; Eeley *et al.* 2001). Forests are furthermore under pressure from rural communities, through collection for use as fuelwood, building materials, food and local medicine (Cocks and Wiersum 2003; Shackleton and Shackleton 2004).

In addition to the threats of deforestation to indigenous forests, habitats have been created in the form of commercial plantations of exotic trees. Plantations have had both positive and negative effects on bird assemblages within forests, and these effects are influenced by factors including tree species used, plantation age, and previous land uses (Bremer and Farley 2010). Plantations can have the positive effects of aiding dispersal of some bird species by acting as corridors between forest patches (Wethered and Lawes 2003, 2005); providing a habitat for species tolerant of plantations (Estades and Temple 1999); and potentially increasing biodiversity if secondary forest or exotic pasture is transformed to plantations (Bremer and Farley 2010). Their negative effects include limiting indigenous forest distribution through the alteration of fire regimes; and limiting the movement of some forest bird species between these fragments of indigenous forest (Geldenhuys 1991; Wethered and Lawes 2003). Studies link increased afforestation through plantations with the replacement of grassland bird assemblages by those traditionally found in wooded habitats, both in South Africa and

globally (Allan *et al*. 1997; Azpiroz *et al.* 2012), and the replacement of grassland, shrubland and indigenous forests with plantations reduces biodiversity (Bremer and Farley 2010). The addition of plantations leads to species assemblages being altered, with few nectarivorous or hole-nesting insectivorous species being found in plantations (Armstrong and van Hensburgen 1995).

In 2009, it was estimated that 10% of South African forest dependent bird species were threatened (Berliner 2009). This number has since doubled, with 19% of forest dependent bird species in South Africa listed as near-threatened or above on the IUCN Red List 2014 (BirdLife South Africa 2014). An understanding of why these changes are occurring is thus essential. The recently updated South African Bird Atlas Project (SABAP), which incorporates volunteer surveying of quarter-degree grid cells from 1987-1992, and then again from 2007 onwards, allows the prospect of investigating changes is avian distribution over the last twenty years. When overlaid with data on changes in land use, including changes in forest distribution and plantations, over the same time period, the relationship between changes in forest distribution and changes in forest dependent bird distribution can be investigated.

The aims of this chapter were: (1) to determine changes in the distribution of forest dependent bird species; (2) to relate these changes to changes in land-use, specifically deforestation of indigenous forests and afforestation with alien plantations; and (3) to identify causal links between these changes, including specific species characteristics and responses.

It was predicted that land use changes would result in deforestation, leading to the decline of forest dependent bird species. It was expected that there would be a mixed response to plantations, with species which thrive in plantations responding negatively to a national loss of plantations, while species which are reliant exclusively on indigenous forests would respond negatively to any increases in plantation cover.

Methods

Species selection and range change

Selected species were listed as forest dependent by Oatley (1989) or as having "high forest dependence" by BirdLife International (2014a). This resulted in a list of 57 forest dependent species. For the full list, see Addendum Table A1. Forest dependency was defined as species which breed almost invariably within forests, depend on forest ecosystems for their ecological requirements, or whose ranges overlap at least 80% with those of forests (Oatley 1989; Lawes *et al.* 2007; Berliner 2009; BirdLife International 2014a).

Species-specific information relevant to the study was determined through analysis of the current literature and was used for characteristics analysis (explained under 'statistical analyses') (Allan *et al*. 1997; Harrison *et al*. 1997; Tarboton 2001; Hockey *et al*. 2005; Chittenden 2007; Hockey and Midgley 2009; Sinclair and Ryan 2009; Sinclair *et al*. 2011; BirdLife International 2014a,b,c; BirdLife South Africa 2014; IUCN Red Data List 2013). This information included data on whether or not a species occurs in plantations, red list status, response to afforestation, forest types the species occurs in, endemicity, where in the forest the species occurs (edge, core, or both), diet, body size and mass, nest type, nest dispersion, nest site fidelity, number of eggs, breeding season, and whether the species was a specialist or a generalist, mobile or sedentary, migratory or resident, gregarious or social, monogamous or polygamous, and precocial or altricial. South African forest types have variously been categorized into between three (Eeley *et al.* 2001), ten (Cooper 1985), twelve (Mucina and Rutherford 2006), fifteen (Acocks 1953) and 23 categories (von Maltitz *et al.* 2003). Here we use the global BirdLife International (2014c) categories, which are based on the IUCN Habitats Classification Scheme (v 3.1), of which six categories (montane, lowland, dry, mangrove, riverine and swamp forest) occur in South Africa (Figure 2.1). These categories were used as information on which habitats bird species utilize were obtained from BirdLife International (2014c), and so the same categories were used throughout this chapter to ensure continuity.

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To determine changes in range size of the 57 species, the South African Bird Atlas Project (SABAP) was used. The first South African Bird Atlas Project (SABAP1), with data collection from 1987-1992, and SABAP2, with data collection from 2007 to September 2014, were compared. The protocol for both comprised volunteer surveying of birds within predetermined grid cells – quarter-degree grid cells were used in SABAP1, while 5 minute by 5 minute pentads were used in SABAP2. Accordingly, for each area covered by a quarter-degree grid cell in SABAP1, nine pentads were used in SABAP2. Comparisons of these datasets are thus possible by combining results from the nine pentads within each quarter-degree grid cell (Harebottle *et al.* 2010), provided that only presence/absence data, rather than those representing reporting rates, are used. Harebottle *et al*. (2010) details further information on data acquisition and validation.
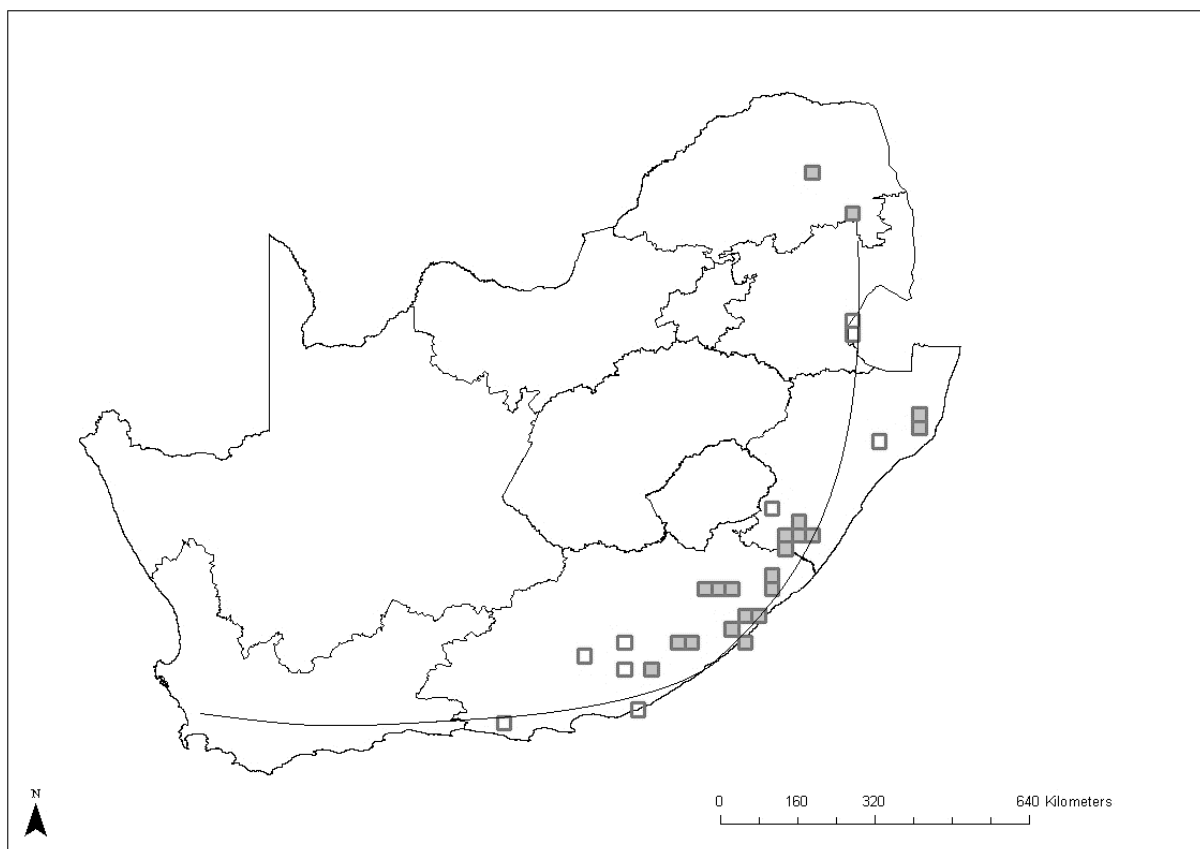
Figure 2.1 Quarter-degree grid cell sites across South Africa that experienced a loss of 20% or more of forest dependent bird species between the two South African Bird Atlas Project (SABAP) periods of 1987-1992 and 2007-present. Shaded sites experienced deforestation of indigenous forest between 2000 and 2013/2014, according to the National Land Cover Dataset (2013/2014) (GeoterraImage Pty Ltd, 2015). Sites not shaded experienced no detectable change in forest cover between 2000 and 2013/2014, but some did between 1994 and 2000 (see text for details). Montane forest occurs above the solid line, and lowland forest below. Other forest types designated by Birdlife International (2014) are not indicated as they are much smaller in extent and are not geographically localized.

Range sizes for each species in SABAP1 and SABAP2 were compared to determine whether the species' ranges were increasing, decreasing or stable. This resulted in a list of 28 decreasing species, 22 increasing species and seven stable species. The larger (and thus coarser) sampling units (quarter-degree grid cells: QDGC) used in SABAP1 compared to the finer scale (pentads) sampling in SABAP2 suggests that species might have been present but not detected in SABAP1, but it is far less likely that species would remain undetected within a given QDGC in SABAP2. Accordingly, a species could falsely be marked "absent" in SABAP1, and then seem to be increasing in SABAP2 when in reality this is a sampling artifact. Therefore, only species with decreasing ranges were used for quantitative analyses in this study (see full list in Addendum Table A1). Percentage range change was used for analyses, and was calculated as the percentage of the range in SABAP1 lost by SABAP2.

Site selection and land cover change

We aimed to identify QDGCs in which more than ten species (>18% of the list of forest dependent species) were present in SABAP 1 but not in SABAP2. This was determined by analysis of each QDGC known to contain either forest or plantation in the last twenty years (van den Berg *et al*. 2008; SANBI 2009; Schoeman *et al*. 2013). Only those QDGC with a sum of four or more SABAP2 report cards were used. Thirty QDGCs met these criteria, 17 of which were situated in the Eastern Cape province of South Africa (Figure 2.1).

Three national land cover datasets were used in this study to determine changes in land cover, supplemented by orthophotos where land cover categorizations were unclear or to differentiate between indigenous forest and plantation cover. The National Land Cover Dataset 1994, hereafter NLC 1994 (Thompson 1999), depicting land cover of the whole country, was used for baseline data. For comparisons of land use cover over time, the National Land Cover 2009, hereafter NLC 2009 (SANBI 2009) was used for KwaZulu-Natal, Mpumalanga and Limpopo provinces, as this was the most recent dataset available at the time of study. The Eastern Cape Land Cover 2014, hereafter EC 2014 (CD: NGI 2010-2012) was used for the Eastern Cape province, as this was the most recent

dataset available at the time. The percentage area covered by each category of land cover within each QDGC was calculated in ArcGIS 10.2 (ESRI 2011). As the different land cover datasets had different categorizations, all data were converted to a coarser scale to be comparable. This dataset comprised five land cover classes, namely cultivation, natural vegetation, plantations, urban infrastructure and waterbodies. The latter category included dams and reservoirs, but not streams or rivers.

In addition to the thirty sites used for range change analyses, 13 of the total of 286 QDGC containing indigenous forest in the country were identified as having no change in species occurrence between SABAP1 and September 2014 of SABAP2. The land use changes of these sites were analysed to gain a better understanding of which specific land use changes lead to a decline in bird species.

The South African National Land Cover Database 2013/2014, hereafter NLC 2014 (GeoterraImage 2014) was released in June 2015. This database could not be used for land cover comparison as the data were released at too late a stage to be incorporated into analyses. This dataset included a category for indigenous forest (but not for forest type), which was excluded from the NLC 2009 and EC 2014. Accordingly, the NLC 2014 was used to determine if the changes in natural vegetation seen between the NLC 1994 and the NLC 2009/EC 2014 were due to a loss of indigenous forest. To accomplish this, data for each QDGC were visually compared with the National Land Cover 2000 (hereafter NLC 2000) (Van den Berg *et al.* 2008) to determine if any changes in indigenous forest extent had occurred between 2000 and 2013/2014. Visual analysis comprised comparison of the land cover categories present in each QDGC for both NLC 2000 and NLC 2013/2014, to ascertain any changes in the extent of land cover classes, and the nature of any changes in land cover (e.g. unimproved grassland to cultivated land). This allowed the determination of any changes in the extent of indigenous forest of all types. This comparison was only valid for the 2000 and 2013/2014 datasets as the differences in categorization of each sequential land cover database made this analysis impossible for the 1994, 2009 and 2014 datasets.

A limitation of the NLC 2000 data is that the 'indigenous forest' class achieved a 42% Users Accuracy (i.e. the probability that a pixel classified into a given category actually represents that category on the ground) (Van den Berg *et al.* 2008). Within the NLC 2013/2014 data the same class achieved a higher (73%) Users Accuracy (GeoterraImage 2014). The accuracy of the NLC 1994 was 79.4% (Fairbanks *et al.* 2000), and so this can be seen as a reasonable good baseline dataset.

Occupancy modelling

Reporting rate was not used as a proxy for abundance in this study due to the inherent flaws in this method when species have low detectability, as with most forest bird species (see MacKenzie *et al.* 2002). Occupancy modelling, using presence/absence data, was used to determine the effects of land cover change on species across the thirty identified QDGSs (sites).

Data for all species were extracted from the SABAP database of the Animal Demography Unit of the University of Cape Town using R (R Core Team 2014). SABAP1 data were extracted from the start of 1 January 1987 to 31 December 1991, and for SABAP2 from 1 July 2007 to 30 September 2014, from the 30 sites. Data formatting was done as per MacKenzie *et al*. (2006).

Single species, multi-season occupancy models were run on PRESENCE (Hines 2006), with SABAP1 as the first season and SABAP2 as the second. Four parameterizations were used to determine best fit, by holding all parameters constant and adding appropriate covariates sequentially for ψ, γ, ε and ρ. These covariates were percentage land cover in 1994 for ψ, and land cover change for γ and ε. ρ was kept constant due to the sampling technique, but seasonal effects were allowed. Additional covariates were then added into a single model to determine best fit. A logistic link was used to calculate probabilities, with 10 000 bootstraps performed. Models with a delta AIC (Akaike's Information Criterion) of less than 2.00 were selected as fitting best. Significance at p = 0.05 was determined using standard errors and 95% confidence intervals.

Statistical analyses

Spearman rank correlations were used to determine correlations among land cover in 1994, and again in 2009 or 2014; and land cover change. Regression analyses were conducted in R to determine relationships between the proportions of species lost in a site and land cover change. T-tests were performed on the number of cards and reports per species and per site to ensure that the numbers for SABAP1 and SABAP2 were comparable.

Species characteristics on all 57 forest dependent species were transformed to a binary matrix for statistical analysis, as per Okes *et al.* (2008). Although some variables could have been recorded as categorical, a binary index was used for all characteristics to allow comparison. Species were categorized by response, as having an increasing range (increasers), having a decreasing range (decreasers), or having a stable range with a change of fewer than two quarter-degree grid squares (stable) before analyses. Data on species characteristics were subsequently grouped by response, and characteristics analyzed as a percentage of the whole to identify patterns. Chi-squared tests for homogeneity were performed to determine significant differences in characteristics among responses, with the hypothesis that species with a similar response to land cover change would exhibit similar characteristics. Chi-square tests were then performed on the characteristics data to determine the prevalence of each category of each characteristic within response groups.

Results

The changes in range size across South Africa for each declining forest dependent species can be seen in Table 2.1. The average change in range size was -16.39%. Species with the largest changes are the rufous-chested sparrowhawk, *Accipiter rufiventris* (-36.33%), the Eurasian golden oriole, *Oriolus oriolus* (-34.62%), and the Cape parrot, *Poicephalus robustus* (-58.33%).

Within the 30 study sites, there were 1225 report cards submitted for SABAP1 (mean 40.83), and 1192 report cards submitted for SABAP2 (mean 39.73). No significant difference was found (p = 0.4678), indicating that the number of cards is comparable. Within the study sites, the number of

Table 1.1 Percentage range change throughout South Africa for all forest dependent bird species which experienced range declines between SABAP1 and SABAP2; as well as the number of report cards for each species across the country; and the number of report cards for each species within the thirty sites which experienced the loss of ten or more forest dependent species. In order of appearance here, the species' name authors are Smith 1830; Daudin 1800; Stephens 1815; Temminck 1809; Linnaeus 1766; Sharpe 1876; Rudebeck 1957; Temminck 1824; Vieillot 1820; Kaup 1850; Lichtenstein 1823; Gmelin 1789; Sundevall 1850; Vieillot 1818; Sharpe 1908; Linnaeus 1766; Linnaeus 1758; Sundevall 1850; Vieillot 1818; Gmelin 1788; Smith 1839; Linnaeus 1766; Smith 1834; Wagler 1827; Shaw 1809; Vieillot 1818; Hartlaub 1864; Vigors 1831.

Scientific name	Common name	Percentage range change	Country-wide		Sites	
			S1	S2	S1	S2
Accipiter rufiventris	Rufous-chested sparrowhawk	-36.33	1227	737	52	5
Accipiter tachiro	African goshawk	-4.69	5907	5369	187	69
Apaloderma narina	Narina trogon	-4.27	2344	1832	123	25
Aplopelia larvata	Lemon dove	-27.72	1275	1185	84	4
Batis capensis	Cape batis	-1.30	6756	10533	297	97
Bradypterus barratti	Barratt's warbler	-4.48	570	819	21	21
Buteo trizonatus	Forest buzzard	-8.05	1571	1893	48	0
Bycanistes bucinator	Trumpeter hornbill	-3.96	6542	4801	163	136
Camaroptera brachyura	Green-backed camaroptera	-13.17	12635	15207	262	367
Circaetus fasciolatus	Southern banded snake-eagle	-16.00	140	186	3	2
Coracina caesia	Grey cuckooshrike	-16.29	1165	1292	54	15
Cossypha dichroa	Chorister robin-chat	-19.53	2710	2556	161	16
Crithagra scotops	Forest canary	-10.23	2376	3204	123	61
Lioptilus nigricapillus	Bush blackcap	-10.71	286	429	10	5
Motacilla clara	Mountain wagtail	-29.61	1794	1223	79	14
Nectarinia chalybea	Southern double-collared sunbird	-12.28	10388	17415	285	60
Oriolus oriolus	Eurasian golden oriole	-34.62	1277	549	21	8
Phylloscopus ruficapilla	Yellow-throated woodland-warbler	-20.69	945	10533	24	18
Pogonocichla stellata	White-starred robin	-23.02	823	701	22	7
Poicephalus robustus	Cape parrot	-58.33	1725	391	43	25
Sarothrura elegans	Buffspotted flufftail	-28.57	954	653	76	12
Stephanoaetus coronatus	African crowned eagle	-14.12	3278	2550	98	28
Strix woodfordii	African wood-owl	-12.32	2958	963	143	22
Tauraco corythaix	Knysna turaco	-4.08	11266	5863	820	58
Telophorus olivaceus	Olive bush-shrike	-2.23	2720	5246	84	67
Trochocerus cyanomelas	Blue-mantled crested-flycatcher	-6.36	1529	2090	75	24
Zoothera gurneyi	Orange ground-thrush	-8.33	197	241	10	0
Zoothera guttata	Spotted ground-thrush	-27.59	185	188	3	2

reports of all declining forest dependent species was 3371 for SABAP1 and 1168 for SABAP2 (p = 0.0096). This decrease in the number of reports (by almost two thirds) is indicative of a true loss of occupancy of these species within these QDGC. Table 2.1 shows a breakdown of the number of report cards submitted for SABAP1 and SABAP2 per species.

Table 2.2 shows the number of report cards submitted for each QDGC which experienced a loss of ten or more bird species between SABAP1 and SABAP2, as well as the number of species lost and gained within each site between SABAP1 and SABAP2. In exactly half of the 30 QDGSs sampling effort was improved or equal in SABAP2 to SABAP1, and the other half it was lower (Table 2.2). In terms of QDGSs which had lost more than ten forest dependent species, losses were most prevalent in the Eastern Cape (n = 17 sites) and KwaZulu-Natal (n = 9 sites). Within the Eastern Cape, 10/17 QDGCs had a decreased sampling effort in SABAP2. Two of these (3225DB and 3326DB) fall outside of the former homelands. Five are in the former Ciskei, all of which had a decreased sampling effort in SABAP2. The remaining three are in the former Transkei, but the remaining six QDGSs in this region had improved sampling effort in SABAP2. Six of the KwaZulu-Natal QDGCs fall within East Griqualand, each of which fell partially in the former Transkei and partially in the former Natal province; again only two (one third) of these sites had a decreased sampling effort in SABAP2, while the other four (two thirds) had improved sampling. Hence, in the former Ciskei reduced sampling effort may have resulted in an overestimation of species loss.

Table 2.2 The number of report cards for each of 30 sites experiencing a loss of 10 or more forest dependent bird species between SABAP1 (1987-1992) and SABAP2 (2007-present), and the number of bird species gains (where a species was not found in a site in SABAP1 but was found there in SABAP2) and bird species losses (where a species was found in a site in SABAP1 but not in SABAP2) from that site between the two atlas periods.

Province	QDGC	SABAP1	SABAP2	Species gains	Species losses
Northern Province	2330CA	76	19	4	11
	2430BD	103	256	3	11
Mpumalanga	2630BD	35	9	0	10
	2630DB	53	5	1	10
KwaZulu-Natal	2831CB	39	45	3	10
	2832AA	48	180	3	10
	2832AC	39	93	2	10
	2929CD	114	186	1	13
	3029BB	50	133	2	15
	3029BC	30	20	3	11

	3029BD	5	22	1	11
	3029DA	20	15	1	11
	3030AC	7	30	4	19
Eastern Cape	3128AC	5	5	0	10
	3128AD	13	6	0	14
	3128BC	3	4	2	13
	3128DD	6	7	2	10
	3129AB	2	5	0	23
	3129AD	1	11	2	11
	3129CC	9	19	2	12
	3225DB	38	5	0	11
	3226BC	8	4	0	18
	3226DC	104	8	0	15
	3227BC	27	7	1	12
	3227BD	8	4	1	10
	3227CC	10	6	1	14
	3228BA	6	5	0	13
	3228BD	17	13	0	10
	3324CD	7	19	0	14
	3326DB	342	51	0	11

Figure 2.2 shows the change in each land use type (cultivation, natural vegetation, urban land use, plantations and waterbodies) between the NLC 1994 and the NLC 2009 or EC 2014 in each of the 30 QDGC that experienced a loss of ten or more bird species between SABAP1 and SABAP2. Natural vegetation has decreased in 20 QDGC, with the largest change being -16.37%. This change in natural vegetation is not limited to indigenous forests, but also includes thickets, grasslands and woodlands. Half of sites experienced an increase in cultivation, with the largest change being 23.64%. Most sites experienced an increase in urban development, and 50% of sites had an increase in plantation cover. Waterbodies (dams and reservoirs) increased in all but one site due to dam construction.
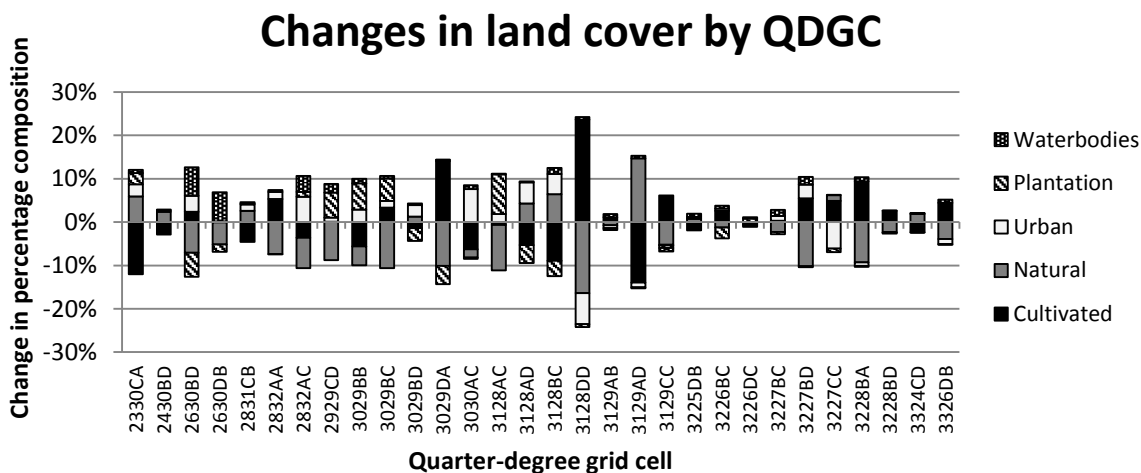
Figure 2.2 Land cover changes in each of the thirty sites that experienced a loss of 20% or more of forest dependent bird species between the two South African Bird Atlas Project (SABAP) periods of 1987-1992 and 2007-present. Natural vegetation is decreasing in most sites, while cultivation and urbanization are increasing. Waterbodies are likewise increasing in most sites, while plantations are increasing in some sites and decreasing in others.

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Figure 2.3 shows the change in each land use type (cultivation, natural vegetation, urban land use, plantations and waterbodies) between the NLC 1994 and the NLC 2009 or EC 2014 in each of the thirteen QDGC that experienced no change in bird species between SABAP1 and SABAP2. Natural vegetation decreased in 11/13 sites, plantations increased in 12/13 sites, cultivation increased in 8/13 sites, urbanization increased in 11/13 sites, and waterbodies increased in 12/13 sites. More of these sites contained an increase in plantations and cultivation than those that experienced a loss of forest dependent bird species (Figure 2.2).

Changes in land cover of QDGC with no change in bird occupancy
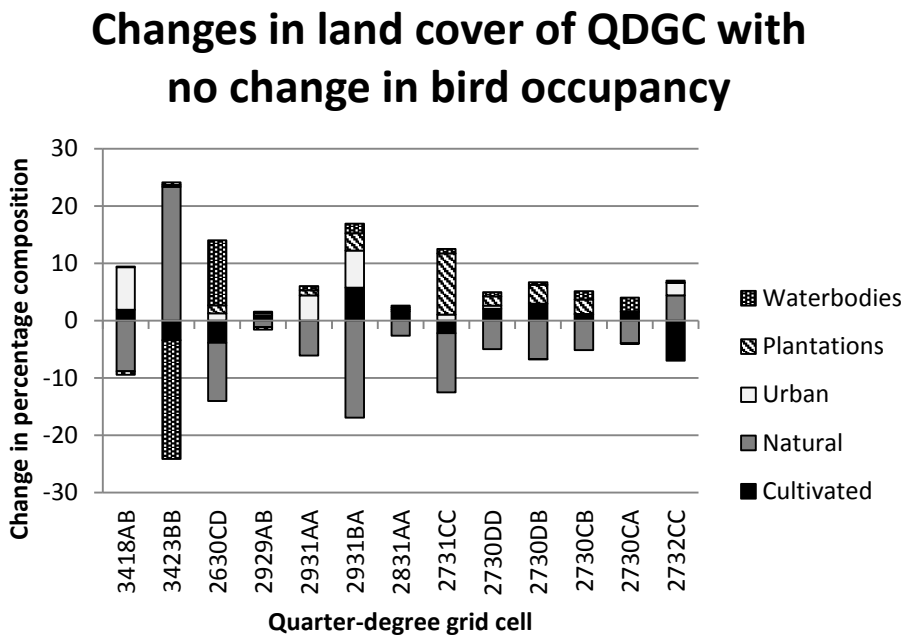
Figure 2.3 Land cover changes in each of the thirteen sites that experienced no change in occupancy of forest dependent bird species between the two South African Bird Atlas Project (SABAP) periods of 1987-1992 and 2007-present.

Table 2.3 summarizes which land cover factors affected the local extinction of each bird species, as well as whether this effect was positive or negative. Changes in plantations increased extinction likelihood in six species (Barratt's warbler *Bradypterus barratti*, bush blackcap *Lioptilus nigricapillus*, Eurasian golden oriole, white-starred robin *Pogonocichla stellata*, blue-mantled crested-flycatcher *Trochocerus cyanomelas* and orange ground thrush *Zoothera gurneyi*), and decreased extinction likelihood in five species (rufous-chested sparrowhawk, Cape batis *Batis capensis*, southern banded snake eagle *Circaetus fasciolatus*, buff-spotted flufftail *Sarothrura elegans* and African wood

owl *Strix woodfordii*). Increased urbanization decreased extinction likelihood in five species (African goshawk *Accipiter tachiro*, Barratt's warbler, bush blackcap, Eurasian golden oriole and white-starred robin), while increases in cultivation decreased extinction likelihood in four species (Barratt's warbler, southern banded snake eagle, bush blackcap and orange ground thrush), and decreases in natural vegetation increased extinction likelihood in three species (Barratt's warbler, Cape parrot and orange ground thrush). Increases in waterbodies increased extinction likelihood in the African goshawk, chorister robin-chat *Cossypha dichroa* and African crowned eagle *Stephanoaetus coronatus*, and mitigated extinction likelihood in Barratt's warbler and the orange ground thrush.

Table 2.3 Summary of significant land cover factors affecting extinction of forest dependent bird species, as well as whether this effect was positive (+) or negative (-). Positive effects indicate that with a larger covariate value the probability of extinction was greater; negative effects indicate that a larger covariate value led to a smaller probability of extinction.

Scientific name	Common name	Factors affecting extinction
Accipiter rufiventris	Rufous-chested sparrowhawk	Plantations -
Accipiter tachiro	African goshawk	Waterbodies +, urbanization -
Batis capensis	Cape batis	Plantations -
Bradypterus barratti	Barratt's warbler	Cultivation -, natural -, urbanization -, plantations +, waterbodies -
Circaetus fasciolatus	Southern banded snake eagle	Plantations -, cultivation -
Cossypha dichroa	Chorister robin-chat	Waterbodies +
Lioptilus nigricapillus	Bush blackcap	Cultivation -, urbanization -, plantations +
Oriolus oriolus	Eurasian golden oriole	Urbanization -, plantations +
Pogonocichla stellata	White-starred robin	Urbanization -, plantations +
Poicephalus robustus	Cape parrot	Natural -
Sarothrura elegans	Buff-spotted flufftail	Plantations -
Stephanoaetus coronatus	Crowned eagle	Waterbodies +
Strix woodfordii	African wood owl	Plantations -
Trochocerus cyanomelas	Blue-mantled crested flycatcher	Plantations +
Zoothera gurneyi	Orange ground thrush	Cultivation -, natural -, urbanization +, plantations +, waterbodies -

Changes in natural vegetation were significantly negatively correlated with changes in cultivation (r_s = -0.7139, n = 29, p < 0.05), as was urban change (r_s = -0.4728, n = 29, p < 0.05). As urbanization increased in almost all sites, this suggests that urbanization is replacing previously cultivated areas. The relationship between natural vegetation and cultivation is less clear, with the possibilities that either natural areas are being cultivated, or cultivated areas are being allowed to return to a more natural state (perhaps associated with a demographic movement into urban areas). The number of species gained per site was significantly positively correlated with the percentage of the site under cultivation in the NLC 1994 (r_s = 0.5735, n = 29, p < 0.05) and the NLC 2009 or EC 2014 (r_s =

0.4336, n = 29, p < 0.05). The number of species lost per site was significantly negatively correlated with the percentage of that site under natural vegetation in the NLC 1994 (r_s = -0.39, n = 29, p < 0.05) and the NLC 2009 or the EC 2014 (r_s = -0.5234, n = 29, p < 0.05).

Species characteristics were analysed within each response group. Chi-squared tests for homogeneity found no significant difference in characteristics among response groups. Pearson's chi-square tests (Table 2.4) showed increasing species to have a significantly higher occurrence in lowland and dry forest (p < 0.005), while decreasing and stable species occurred most frequently in montane and lowland forest (p < 0.005 in both cases). More decreasing than increasing species were monogamous (p < 0.005). Solitary nest dispersion was prevalent in both increasing (p = 0.05) and decreasing species (p < 0.005). A higher proportion of decreasing species have built nests (p < 0.005) than other nest categories. A higher proportion of stable species were insectivores (p = 0.025). Stable species also had a significantly lower body size (<20cm, p = 0.025) and body mass (<100g, p = 0.025), and have a tendency to breed in summer (p = 0.025).

Table 2.4 Results of the Chi-square test on species characteristics of South African forest dependent birds. Significance is marked at the 0.05 (*), 0.025 (**) and <0.005 (***) level.

Characteristic	Increasing	Decreasing	Stable
Plantation occurrence	0.995	0.99	0.9
Forest dependency	0.995	0.995	0.9
Threatened	0.995	0.995	0.9
Response to afforestation	0.95	0.995	0.9
Forest type	<0.005***	<0.005***	<0.005***
Specialist/Generalist	0.995	0.995	0.995
Mobile/Sedentary	0.99	0.995	0.9
Migrant/Resident	0.9	0.9	0.95
Endemicity	0.95	0.99	0.9
Gregarious/Solitary	0.995	0.995	0.975
Location in forest	0.995	0.995	0.995
Diet level 1	0.995	0.975	0.9
Diet level 2	0.9	0.9	0.025**
Body size (cm)	0.9	0.95	0.025**
Nest type	0.1	<0.005***	0.1
Breeding system	0.9	<0.005***	0.9
Nest dispersion	0.05*	<0.005***	0.9
Nest site fidelity	0.995	0.995	0.975
Body mass (g)	0.9	0.9	<0.005***
Precocial/Altricial	0.9	0.9	0.9
Number of eggs	0.995	0.975	0.95
Breeding season length	0.95	0.99	0.9
Breeding season	0.9	0.9	0.025**

Figure 2.1 indicates which sites exhibited a loss of indigenous forest between the NLC 2000 and NLC 2014. Of the thirty sites analysed in this study, 22 were found to have experienced deforestation between the NLC 2000 and the NLC 2014 (these are shaded in Figure 2.1). Fifteen of these deforested sites fell within the 20 QDGC in which natural vegetation decreased between the NLC 1994 and the NLC 2009/EC 2014 (Figure 2.2), so the changes in these fifteen sites can be attributed to loss of indigenous forests. The seven sites remaining (Figure 2.2) of the 22 experiencing deforestation between the NLC 2000 and the NLC 2014 increased in natural vegetation between the NLC 1994 and the NLC 2009/EC 2014.

Of the thirty sites analysed in this study, eight were found to exhibit no detectable change in forest cover between the NLC 2000 and the NLC 2014 (these are unshaded in Figure 2.1). Five of these experienced a decrease in natural vegetation between the NLC 1994 and the NLC 2009/EC 2014 (Figure 2.2). Four of the five sites exhibited a loss of indigenous forest between the NLC 1994 and the NLC 2000, while all experienced loss and fragmentation of plantations, including clear-felling. The remaining three sites did not exhibit a detectable change in forest extent between the NLC 2000 and the NLC 2014, but two had a decrease in plantation cover prior to 2000. All three had an increase in natural vegetation between the NLC 1994 and the NLC 2009/EC 2014.

Discussion

The results of this study suggest that at least 50% of forest dependent birds in South Africa are experiencing range declines (Table 2.1). In terms of QDGSs which had lost more than ten forest dependent species, losses were most prevalent in the Eastern Cape (n = 17 sites) and KwaZulu-Natal (n = 9 sites). The forests of the former homelands were transferred from the former Ciskei and Transkei forestry departments to the national forestry department post-1994. The inland forests of the Eastern Cape and former East Griqualand (the latter now forms part of the province of KwaZulu-Natal) are associated with plantations, granting them some measure of protection as a result of this proximity. However, the majority of the coastal forests (most of which fall within the former Transkei) are not associated with plantations, have not been effectively policed post 1994, resulting in

alien invasion, deforestation, and some illegal harvesting of trees (pers. comm. J. Feely). This may lead to a difference in the changes in land cover experienced by these sites, and a consequent difference in the response of species to these changes. The most recent data on indigenous forest cover show in particular deforestation since 2000, primarily in the Eastern Cape (Figure 2.1).

Extinction of forest dependent birds was influenced primarily by plantation cover (Table 2.3), which increased in 50% of sites, replacing thickets, grassland and indigenous forest (Thompson 1999; GeoterraImage 2014). Plantations are unsuitable habitats for species that build nests in the undergrowth; Barratt's warbler, bush blackcap, white-starred robin, blue-mantled crested-flycatcher and orange ground thrush all fall into this category (Tarboton 2001). The Eurasian golden oriole, a non-breeding migrant, is also negatively affected by planation cover, despite being known to occur in plantations elsewhere in the world (e.g. tea plantations and palm plantations in India; Sinu 2011; Basheer and Aarif 2013). Plantation cover contributed towards extinction in the above six species but mitigated it in five other forest dependent bird species which appear to benefit from plantations. Birds of prey have long been known to utilize plantations for nesting and feeding (Prestt 1965), explaining why the rufous-chested sparrowhawk, southern banded snake eagle and African wood owl were found to benefit from plantations. Two additional species appeared to benefit from increased plantation cover: the Cape batis which is known to forage within plantations in South Africa (Earlé and Oatley 1983; Armstrong and van Hensbergen 1994) and the buff-spotted flufftail, also often found in plantations (BirdLife International 2014c).

Urbanization and cultivation influenced extinction to lesser extents. Urban gardens can act as refuges for forest birds, and frugivores in particular are found in urban areas (Biggs *et al.* 2006). In five species (African goshawk, Barratt's warbler, Bush blackcap, Eurasian golden oriole and white-starred robin), urbanization had the effect of mitigating extinction possibility, and in only a single species – the orange ground thrush, did urbanization increase extinction. The number of species gained per site increases with the percentage of the site under cultivation, indicating that some species, at least, adapt to cultivated areas. Woodland birds have been known to venture onto cultivated land to feed and

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breed in hedgerows in Europe (Wilson *et al.* 1999; Hinsley and Bellamy 2000), and forest birds in Costa Rica forage in agricultural landscapes (Hughes *et al.* 2002). The Cape parrot feeds in fruit orchards in the area surrounding indigenous forests in South Africa (Wirminghaus *et al.* 2001a; Downs 2005). In four South African forest species (Barratt's warbler, Bush blackcap, orange ground thrush and southern banded snake eagle) extinction was mitigated by cultivation, suggesting that these species use farmland adjacent to forests in which to feed.

Decreases in natural vegetation contributed directly towards extinction likelihood in three species (Barratt's warbler, Cape parrot and orange ground thrush). The number of species lost per site (Table 2.2) increased with the extent of loss of natural vegetation in that site, Deforestation of indigenous forests was found in 17% of sites between 1994 and 2000; and in over 70% of sites between 2000 and 2013/2014 (Figure 2.1). Loss of plantations in some sites (Figure 2.2) has also contributed to species loss, as more than 50% of forest dependant birds with decreasing ranges occur in plantations (see Addendum Table A2).

Raptor deaths are attributed to drowning in waterbodies (Anderson *et al.* 1999; Anderson 2000); this explains the increase in extinction likelihood with increased waterbodies of the African goshawk and crowned eagle. The orange ground thrush is known to favour stream edges in forest (Sinclair *et al.* 2011), and could therefore be attracted by waterbodies if sufficient tree cover was present, although it is unclear why the chorister robin-chat and Barratt's warbler are negatively and positively affected by waterbodies, respectively.

When sites across the country which had experienced no change in forest dependent birds were examined (Figure 2.3), it was found that these sites were more likely to have increased cultivation and plantations than sites which had lost bird species (Figure 2.2). The buffering effects of plantations and cultivated areas for forest dependent birds could explain the range stability of the species found in these sites.

The species found to be experiencing the greatest loss in range were the Eurasian golden oriole, red-breasted sparrowhawk and the Cape Parrot. It is important to note that range declines do not necessarily correspond to population declines. A study by Downs *et al.* (2014) on the long-term population trends of the Cape Parrot in South Africa found that, while the proportion of locations in which Cape Parrots were observed decreased over a fifteen year period, the abundance of the species increased. These data were not included in the SABAP2 data used for the present study, perhaps explaining the disparity in results. The Cape Parrot is large and mobile, and frequently forages long-distance. This foraging occurs in flocks (Wirminghaus 2002), perhaps explaining why the species is able to occur in large numbers, as found by Downs *et al.* (2014).

The occurrence of 37/57 species in montane and 45/57 species in lowland forests (categories as per the BirdLife International (2014) list of habitat categories) suggests that these forest subtypes should enjoy the highest conservation priority. This is especially true for montane forests, as most decreasing species occur here (refer to Addendum Table A2), indicating that this vegetation type is most at risk. If forests are degraded, or cleared, the lack of nest building material may restrict reproduction of decreasing species with built nests (Addendum Table A2), as the abundance of vegetation utilized in nest-building is known to affect nesting success, for example in species utilizing epiphytes in coffee plantations in Mexico (Cruz-Angón and Greenberg 2005), and utilizing grasses in grasslands and pastures in Illinois, U.S. (Roseberry and Klimstra 1970). Small, insectivorous species, like the majority of "stable" species in this study, can presumably survive in smaller forest patches, as has been found in the Andes (Kattan *et al.* 1994).

Maintaining the diversity of species guilds present in a natural environment is vital to the functional processes of ecosystems. Healthy plant populations are maintained by insectivorous birds through insect predation, and this guild is more prominent in heterogeneous forests (Sekercioglu 2010; Bereczki *et al.* 2014). Cavity nesters are the most important of these insectivores, and are the first to disappear from exploited forests, with the removal of dead wood changing resource availability (Du Plessis 1995; Bereczki *et al.* 2014). Likewise, forest regeneration through the plant-frugivore network,

can be affected by a loss of dispersers reducing tree recruitment (Cordeiro and Howe 2001; Chama *et al.* 2013b). Frugivores generally subsist on only a subset of the fruiting species available, and therefore conserving forest heterogeneity and fragment size is important for their persistence (Cordeiro and Howe 2001; Bleher *et al.* 2003). Frugivores can also be affected indirectly, as pollination by bird species is restricted in a fragmented landscape, which can lead to lower fruit sets and thus limit frugivore food sources (Cunningham 2000). A loss of frugivores in a community will inevitably lead to the vulnerability of more specialized plant species, potentially altering species richness (Chama *et al.* 2013b). Where forest fragments do possess a high amount of fruit availability, they can be instrumental in maintaining the connectivity of forest fragments and patches in a matrix, if the forest community is one tolerant of fragmentation (Berens *et al.* 2014). This resource availability is a crucial determinant in the health of the plant-frugivore network (Chama *et al.* 2013b).

Seed dispersal is recognized as one of the most important ecological functions of birds, and loss of forest habitats has been linked to losses of bird dispersers and resultant lower tree recruitment (Howe and Smallwood 1982; Cordeiro and Howe 2001). The Cape parrot is dependent on *Podocarpus* spp (yellowwood tree species) for both food and reproduction, nesting in holes in large yellowwood trees often utilized in logging (Wirminghaus *et al.* 2001b; Downs 2005). As *Podocarpus* species are dispersed by birds (Adie and Lawes 2011), a reduction in trees due to logging and deforestation would lead to a reduction in the bird species dependent on them, which would in turn limit dispersal of remaining trees. Hornbills, such as the trumpeter hornbill documented here to be undergoing a range decline, are keystone species within forests and vital for seed dispersal (Trail 2007). Trumpeter hornbills have been found to be important dispersers of seeds within and between South African forest patches, where seed removal rates decline with increasing degradation of forests and deforestation (Kirika *et al.* 2008; Lenz *et al.* 2011).

The bird diversity of South African forests is under threat from anthropogenic land use change and deforestation of indigenous forests, although some relief may lie in unexpected land uses such as

plantations, urban gardens and cultivated areas. Range declines in forest dependent species will be arrested only through active efforts to conserve the remaining South African forest fragments.

Chapter 3 - A species assessment of declining South African forest dependent birds

Cooper, TJG[1]; Wannenburgh, A[2]; Cherry, MI[1]

[1]Department of Botany and Zoology, Stellenbosch University; [2]Department of Environmental Affairs, South Africa

Abstract

Forests in South Africa have had a long history of human utilization, resulting in deforestation and forest degradation. This has had an impact on the biota native to these forests which often differs depending on species-specific characteristics. Twenty eight forest dependent birds with decreasing ranges between 1992 and 2014 were examined to determine links between land use change, characteristics and range declines in 30 quarter-degree grid squares across South Africa where more than ten forest dependent bird species were lost during this period. Most (22) of these sites were in the Eastern Cape or in the East Griqualand, formerly part of the Transkei homeland. Most species lost were birds of prey or insectivores, and the Cape Parrot (*Poicephalus robustus*), rufous-chested sparrowhawk (*Accipiter rufiventris*) and the migratory Eurasian golden oriole (*Oriolus oriolus* suffered the largest declines in range size. The main cause of loss of forest dependent species was a loss of indigenous forest, which has been aggravated in half of these species by recent declines in plantation forests.

Introduction

The naturally patchy indigenous forests of South Africa contain some of the highest species richness of any temperate forests (Low and Rebelo 1996; Silander 2000) and contain 14% of the country's terrestrial bird species (Geldenhuys and MacDevette 1989). Species occurrence patterns within these forests are thought to be a result of paleoclimatic changes during the Quaternary; extinction events resulting from these changes in climate and, more recently, anthropogenic fragmentation and deforestation (Lawes *et al.* 2004a). Forest resources in South Africa began to be heavily utilized with colonial settlement in the 1800s (von Maltitz and Shackleton 2004). This exploitation included logging, clearing for agriculture and increased frequency of fires (Berliner 2009). In addition, grassland and fynbos adjacent to indigenous forests, and occasionally the forests themselves, were

cleared and planted with plantations of exotic trees from as early as 1887 (von Maltitz and& Shackleton, 2004). Current estimates on the degree of indigenous forest loss are unreliable, due to the lack of data available for historic forest extent (Berliner 2009). Furthermore, a large proportion of remaining forest is thought to be degraded, with some loss of ecosystem functioning (Berliner 2009).

The local and landscape-level changes which occur in the matrix surrounding forest patches are considered one of the most important threats facing forest ecosystems in South Africa, with one third of the forest matrix estimated to be affected (Berliner 2009). A patch of indigenous forest in a matrix containing plantations will have a different species complex than one in a matrix containing only natural vegetation (Wethered and Lawes 2005), as this matrix influences the movement of species between forest patches (Wethered and Lawes 2003). Plantations may act as a corridor and extinction buffer for some species within small forest fragments, by increasing the likelihood of their colonisation and immigration (Wethered and Lawes 2003). However, plantations can also act as a barrier to other species, limiting their dispersal (Geldenhuys 1991; Wethered and Lawes 2003). There has been a loss of plantations nationally over recent years, with a decrease of 0.9% per annum between 1999 and 2009, although the Eastern Cape province was an exception - it experienced a 4.3% growth in plantations between 2002 and 2008 (DWAF 2011). Plantation loss could negatively impact those species utilizing plantations, or even those in indigenous forest fragments linked by plantations.

There are differential responses to habitat disturbance and loss among the species found within forests, with some faring better than others (Ewers and Didham 2006; Lawes *et al.* 2006; Wilson *et al.* 2007; Berliner 2009). In tropical forests, the effects of habitat fragmentation on bird extinction may continue long after the disturbance has occurred (Brooks *et al.* 1999). Species-specific habitat requirements and characteristics influence species abundance (Banks *et al.* 2012) and response to fragmentation (Ewers and Didham 2006). In addition, local parameters such as fragment size and shape and the complexity of the surrounding matrix have an impact on species persistence (Ewers and Didham 2006).

Species-specific characteristics influencing a species' response to deforestation and afforestation with plantations include mobility (Hinsley *et al.* 2009), degree of specialization (Ewers and Didham 2006; Hinsley *et al.* 2009), trophic level and body size (Schoener 1968; Ewers and Didham 2006). Synergistic effects resulting from habitat alteration, such as changes in pollination and increased incidence of disease, also influence species response to habitat loss (Ewers and Didham 2006). The literature suggests that the differential response of species to habitat loss should lead to patterns in the suites of species lost. This study examines the forest dependent bird species lost in thirty sites across South Africa over a twenty-two year period between 1992 and 2014. Here we examine each of the 28 bird species suffering range declines in an attempt to determine the extent, location and causes of each decline. It was predicted that species which prosper in plantations will respond negatively to a loss in plantations, while those that rely exclusively on indigenous forests will respond negatively to any increase in plantations.

Methods

Species selection was based on Oatley's list of forest dependent birds (Oatley 1989) and BirdLife International's list of birds with "high forest dependence" (BirdLife International 2014a), resulting in a list of 57 species. When range changes for these species were analysed by comparison of data from the South African Bird Atlas Project (SABAP) 1 and 2 (data from 1987-1992 and 2007-September 2014, respectively), 28 species were found to have decreasing ranges. Decreasing ranges were classed as those where total range size was reduced by two or more quarter-degree grid cells between the two SABAP periods. Only quarter-degree grid cells within South Africa, Lesotho and Swaziland were considered. Harebottle *et al*. (2010) detail further information on data acquisition and validation. The larger (and thus coarser) sampling units (quarter-degree grid cells) used in SABAP1 compared with the finer scale (pentads) sampling in SABAP2 suggests that species might have been present but not detected in SABAP1, but far less likely that species would remain undetected within a given QDGC in SABAP2. Accordingly, a species could falsely be marked "absent" in SABAP1, and then seem to be increasing in SABAP2 when in reality this is a sampling artefact. Accordingly, the results of this study are conservative.

Species-specific information, including red list status (IUCN Red Data List 2013); habitat (BirdLife International 2014a; Sinclair *et al.* 2011); whether the species is found in plantations (BirdLife International 2014a); diet (Hockey *et al.* 2005; Sinclair *et al.* 2011); and whether a species is migratory or resident (Sinclair *et al.* 2011), was determined from the literature.

Sites selected were quarter-degree grid cells (QDGC) where more than ten bird species (~18% of the list of forest dependent species) were lost between SABAP1 and SABAP2. Sites further had a minimum of report cards of four for SABAP2, as this is considered a minimum sampling size (pers. comm. Assoc. Prof. Res Altwegg). This resulted in thirty sites (see Addendum Figure A1), in which changes in species occupancy and land cover were analysed in this study.

To determine changes in land cover in these sites, data on national land cover from 1994, 2000, 2009 and 2013/2014 were analysed visually and by percentage cover of each land cover class. These data were from the National Land Cover Dataset 1994 (Thompson 1999), the National Land Cover 2000 (hereafter NLC 2000) (Van den Berg *et al.* 2008), the National Land Cover 2009 (for KwaZulu-Natal, Mpumalanga and Limpopo provinces) (SANBI 2009), the Eastern Cape Land Cover 2014 (for the Eastern Cape province) (CD: NGI 2010-2012), the South African National Land Cover Database (2013/2014) (GeoterraImage 2014). The percentage area covered by each category of land cover within each QDGC was calculated in ArcGIS 10.2 (ESRI 2011). As the different land cover datasets had different categorizations, all data were converted to a coarser scale to be comparable. This dataset comprised five land cover classes, namely cultivation, natural vegetation, plantations, urban infrastructure and waterbodies. The latter category included dams and reservoirs, but not streams or rivers.

Visual analysis comprised comparison of the land cover categories present in each QDGC for both NLC 2000 and NLC 2013/2014, to ascertain any changes in the extent of land cover classes, and the nature of any changes in land cover (e.g. unimproved grassland to cultivated land). This allowed the determination of any changes in indigenous forest extent. This comparison was only valid for the

2000 and 2013/2014 datasets as the differences in categorization of each sequential land cover database made this analysis impossible for the 1994, 2009 and 2014 datasets. This comparison allowed the allocation of a 'deforestation' category (Table 3.3), but it was not possible to calculate the percentage of forest loss due to the problems outlined above.

Occupancy modelling was done per bird species to determine the effects of changes in land cover on each species. Data from the SABAP database were used in conjunction with data on percentage land cover by class from the National Land Cover 1994 (Thompson 1999), the National Land Cover 2009 (SANBI 2009), and the Eastern Cape Land Cover 2014 (CD: NGI 2010-2012) to construct single-species, multi-season models. Percentage land cover for each class (cultivation, natural, urban, plantations and waterbodies) in 1994 was used as a covariate to determine initial range changes (ψ). The difference in percentage land cover between 1994 and 2009/2014 for each class was used as a covariate to determine colonisation (γ) and extinction (ε). Four parameterizations were used to determine best fit, by holding all parameters constant and adding appropriate covariates sequentially for ψ, γ and ε. Probability of observation (ρ) was kept constant due to the sampling techniques used in SABAP1 and SABAP2, but seasonal effects were allowed. Once the best parameterization was ascertained, additional covariates were added into a single composite model to determine best fit. A logistic link was used to calculate probabilities, with 10 000 bootstraps performed. Models with a delta AIC (Akaike's Information Criterion) of less than 2.00 were selected as fitting best.

The methodology used to determine changes in land cover and to model occupancy was the same as was followed for chapter 2. In chapter 3 species characteristics and habitat requirements, as well as the history of the sites they were lost from, were analysed in more detail. The characteristics, range change and ecology of each species were analysed to determine any links to changes in land cover or site-specific changes. An extensive literature review of each species was done, to further understand patterns in species loss.

Results

The thirty sites analysed in this study, and the forest dependent bird species which were lost from each, can be seen in Table 3.1.

Table 3.1 Forest-dependent bird species changes per quarter-degree grid cell in South Africa between SABAP1 (1987-1992) and SABAP2 (collected from 2007-September 2014). "I" indicates an increase; "D" a decrease, and a blank cell indicates no change.

Grid	Rufous-chested sparrowhawk *Accipiter rufiventris*	African goshawk *Accipiter tachiro*	Narina trogon *Apaloderma narina*	Lemon dove *Aplopelia larvata*	Cape batis *Batis capensis*	Barratt's warbler *Bradypterus barratti*	Forest buzzard *Buteo trizonatus*	Trumpeter hornbill *Bycanistes bucinator*	Green-backed camaroptera *Camaroptera brachyura*	Southern banded snake-eagle *Circaetus fasciolatus*	Grey cuckooshrike *Coracina caesia*	Chorister robin-chat *Cossypha dichroa*	Forest canary *Crithagra scotops*	Bush blackcap *Lioptilus nigricapillus*	Mountain wagtail *Motacilla clara*	Southern double-collared sunbird *Nectarinia chalybea*	Eurasian golden oriole *Oriolus oriolus*	Yellow-throated woodland-warbler *Phylloscopus ruficapilla*	White-starred robin *Pogonocichla stellata*	Cape parrot *Poicephalus robustus*	Buffspotted flufftail *Sarothrura elegans*	African crowned eagle *Stephanoaetus coronatus*	African wood owl *Strix woodfordii*	Knysna turaco *Tauraco corythaix*	Olive bush-shrike *Telophorus olivaceus*	Blue-mantled crested-flycatcher *Trochocerus cyanomelas*	Orange ground-thrush *Zoothera gurneyi*	Spotted ground-thrush *Zoothera guttata*
2330CA		D				D	D					D				D		I		D		D					D	
2430BD			D	D		D	D				D	D										D			D			
2630BD	D	D		D			D		D			D			D									D	D			
2630DB	D	D	D	D								D			D									D	D			
2831CB			D				D				D	D	D			D								D		D		
2832AA				I		D	D			D	D								D									
2832AC			D																		D	D				D	D	D
2929CD		D	D			I	D		D			D	D	D					D			D		D				
3029BB			D					D				D	D		D	D			D	D	D	D					D	D
3029BC			D									D							D	D		D			D	D	D	D
3029BD	D	D	D	D			D													D		D	D					D
3029DA	D		D				D	D							D							D	D					D
3030AC	D		D	D		D	I	D			D	D			D	I						D					I	
3128AC					D				D			D	D	D		D		D						D				
3128AD					D				D		D	D		D		D				D		D		D	D			
3128BC		D	D	D			I					D	D			D	D		D	D		D						
3128DD		D		D				I				D	D						D			D						
3129AB		D	D	D	D				D	D		D		D		D	D		D	D	D	D		D			D	D
3129AD			D										D	D	I	D			D	D		I				D		
3129CC		D	D				D					D		D					D									
3225DB	D	D		D								D										D						
3226BC	D	D	D	D								D				D				D		D			D	D		
3226DC	D	D		D	D							D							D			D						
3227BC	D	D	D					D				D	D			D						D						
3227BD		D		D				D												D		D		D		D		
3227CC		D	D	D	D			D				D	D			D					D	D	D		D			
3228BA		D	D		D							D				D			D	D		D		D	D	D		
3228BD			D	D																	D	D	D					D
3324CD	D	D							D			D	D						D			D	D		D			
3326DB		D		D			D	D					D			D			D			D						

Occupancy modelling gave an indication of which land cover changes affect species presence/absence, although other effects might also be occurring (Table 3.2). "Likelihood of presence" gives an indication of which land cover factors impact distribution, with a positive effect indicating a land cover type which is a determinant of initial distribution. "Likelihood of extinction" gives an indication of which land cover changes influence extinction. The extent of waterbodies was significant in determining the initial ranges of sixteen of the 28 species (Table 3.2). Waterbodies increased in extent – up to a maximum of seven per cent - at all but two of the thirty sites, while plantations increased in 19 sites (up to a maximum of nine per cent), remained the same in two, and decreased (up to a maximum of six per cent) in 11 sites (Table 3.3). Six species went extinct in areas with increased plantations, while five went extinct in areas which lost plantations (Table 3.2). Five species benefitted from increased urbanization, while one went extinct in an area with increased urbanization (Table 3.2).

Table 3.2 Results of the occupancy modelling per forest dependent bird species, where the land cover category is indicated by C (cultivation), N (natural vegetation), P (plantations), U (urban) and W (waterbodies), and the effect of increases in percentage land cover on species presence or extinction is positive (+), negative (-), or has no effect (blank).

Species	Likelihood of presence					Likelihood of extinction				
	C	N	P	U	W	C	N	P	U	W
Rufous-chested sparrowhawk	-				+		-			
African goshawk			+		+				-	+
Narina trogon	+		+		+					
Lemon dove		-		-	+					
Cape batis			+				-			
Barratt's warbler	-	-	-	+	+	-	-	+	-	-
Forest buzzard		-		-	-					
Trumpeter hornbill					+					
Green-backed camaroptera			+		+					
Southern banded snake-eagle				-		-		-		
Grey cuckooshrike					+					
Chorister robin-chat										+
Forest canary					+					
Bush blackcap						-		+	-	
Mountain wagtail			+		+					
Southern double-collared sunbird		+			+					
Eurasian golden oriole	-							+	-	
Yellow-throated woodland-warbler					+					
White-starred robin	-							+	-	
Cape parrot				-		-				
Buffspotted flufftail							-			
African crowned eagle										+
African wood owl	-				+	-				
Knysna turaco		+								
Olive bush-shrike			+		+					
Blue-mantled crested-flycatcher	-				+			+		
Orange ground-thrush	-				-	-	-	+	+	-
Spotted ground-thrush					+					

Sites were analysed to determine the percentage change of each land cover category between 1994 and 2009/2014. The results of this analysis can be seen in Table 3.3. Further visual analyses of the land cover data for these sites was undertaken to determine specifics of land cover change; the results of these analyses can be seen in Table 3.4.

Table 2 Changes in percentage land cover of each site, and deforestation of indigenous forest between 1994 and 2014. The province in which the quarter-degree grid cell occurs is indicated in the first column, with the Northern Province (NP), Mpumalanga (MP), KwaZulu-Natal (KZN) and the Eastern Cape (EC) represented.

	QDGC	Cultivation	Natural vegetation	Plantations	Urbanization	Waterbodies	Deforestation
NP	2330CA	-12.06%	5.88%	2.58%	2.82%	0.78%	Y
	2430BD	-2.88%	2.36%	0.00%	0.41%	0.12%	Y
MP	2630BD	2.38%	-7.07%	-5.53%	3.70%	6.51%	
	2630DB	-0.10%	-5.03%	-1.75%	0.14%	6.74%	
KZN	2831CB	-4.60%	2.58%	0.25%	1.47%	0.30%	
	2832AA	5.29%	-7.40%	-0.02%	1.64%	0.49%	Y
	2832AC	-3.58%	-7.08%	1.05%	5.86%	3.75%	Y
	2929CD	0.07%	-8.79%	5.73%	0.96%	2.03%	
	3029BB	-5.59%	-4.38%	6.05%	2.89%	1.03%	Y
	3029BC	3.34%	-10.64%	5.20%	1.52%	0.59%	Y
	3029BD	-1.36%	1.23%	-2.95%	2.67%	0.42%	Y
	3029DA	14.05%	-10.12%	-4.22%	0.24%	0.05%	Y
	3030AC	-6.27%	-1.92%	-0.33%	7.57%	0.94%	Y
EC	3128AC	-0.65%	-10.50%	9.11%	1.87%	0.17%	Y
	3128AD	-5.27%	4.29%	-4.19%	4.87%	0.20%	Y
	3128BC	-8.89%	6.41%	-3.58%	4.67%	1.39%	Y
	3128DD	23.64%	-16.37%	-0.68%	-7.14%	0.55%	Y
	3129AB	1.10%	-0.74%	-0.38%	-0.69%	0.72%	Y
	3129AD	-13.92%	14.66%	-0.24%	-1.12%	0.61%	Y
	3129CC	6.15%	-5.27%	-0.59%	-0.11%	-0.80%	Y
	3225DB	-1.92%	0.71%	0.00%	0.49%	0.72%	Y
	3226BC	2.61%	-1.28%	-2.49%	0.49%	0.67%	Y
	3226DC	-0.42%	-0.24%	0.80%	-0.44%	0.30%	Y
	3227BC	0.32%	-2.32%	-0.47%	1.05%	1.42%	Y
	3227BD	5.48%	-10.18%	-0.23%	3.18%	1.74%	Y
	3227CC	4.89%	1.28%	-0.81%	-6.10%	0.05%	Y
	3228BA	9.40%	-9.28%	-0.18%	-0.86%	0.92%	Y
	3228BD	2.58%	-2.35%	-0.07%	0.05%	-0.21%	Y
	3324CD	-2.37%	1.85%	0.14%	-0.01%	0.09%	Y
	3326DB	4.43%	-3.89%	-0.14%	-1.18%	0.78%	Y

Deforestation was analysed with a visual examination of land cover datasets, and was not ground-truthed.

Table 3.4 Results of the visual analysis of the changes in land cover of the thirty sites with a loss of ten or more forest dependent birds between 1994 and 2009/2014. These data are the result of land cover data only; no ground trothing was done due to time and cost limitations.

QDGC	Changes
2330CA	Degradation of thicket and bushland; plantation in thicket and bushland; urbanization; plantation up to forest border; plantation in former forest; cultivation of former degraded forest; riparian invasion
2430BD	Degradation and fragmentation of forest; cultivation up to forest border; riparian invasion; cultivation in previous forest; deforestation
2630BD	Urban and cultivation increase; riparian invasion
2630DB	Plantations up to plantation borders; reduced corridors and grassland mosaic; riparian invasion
2831CB	Fragmentation and reduction of plantations; cultivation of grasslands; cultivated to natural and plantation; urbanization; riparian invasion
2832AA	Cultivation of grassland and thicket and bushland; cultivation to natural vegetation; degradation of thicket and bushland; deforestation of forest and woodland; riparian invasion
2832AC	Urbanization; cultivation of forest and thicket and bushland; cultivation to natural vegetation and plantation; grassland and thicket and bushland to plantations; new plantations at former plantation edge; riparian invasion
2929CD	Urbanization; cultivation of grasslands; fragmentation of natural vegetation; increase in plantations; new plantations up to former plantation border; riparian invasion
3029BB	Cultivation of grassland; urbanization; cultivation to degraded natural vegetation; increase in waterbodies; plantations at forest borders; plantations in grassland; plantations in thicket; new plantations up to former plantation borders; urbanization; riparian invasion
3029BC	Plantations in previous forest and thicket and bushland and grassland; plantations at forest borders; new plantations at former plantation borders; fragmentation of plantations; increased urbanization, cultivation and waterbodies; riparian invasion
3029BD	Cultivation of degraded and improved grassland; urbanization; massive plantation removal; forest removal; riparian invasion
3029DA	Plantation surrounding forest; fragmentation of forest and plantation; plantation in former grassland and forest; cultivation in former grassland; urbanization; riparian invasion
3030AC	Plantation borders cultivated (previously grassland); urbanization; plantation in former grassland; fragmentation of plantations; riparian invasion; increased waterbodies
3128AC	Plantation in former grassland; massive afforestation with plantations; fragmentation of indigenous forests; urbanization; riparian invasion
3128AD	Reduction in plantation and forest; plantation up to forest border; clearing of plantation for pasture; urbanization; riparian invasion
3128BC	Alien invasion at plantation edges; fragmentation of plantations and forest; bush encroachment of grassland; urbanization; riparian invasion
3128DD	Alien invasion; deforestation of forests and plantations; fragmentation of forests; riparian invasion; urbanization
3129AB	Plantation in former forest; mass scale invasion
3129AD	Mass scale invasion; possible forest expansion; loss of plantations
3129CC	Replacement of plantations with indigenous forests; fragmentation of forests; riparian invasion
3225DB	Huge loss of forest and woodland to grassland; some new forest in kloofs; invasion; loss of plantations
3226BC	Forests in former plantation; mass-scale invasion
3226DC	Increase in plantations and forest cover; invasion
3227BC	Huge loss of forest and woodland; loss of smaller plantation patches; invasion
3227BD	Reduction in plantation patch size; loss of forest patch; invasion; urbanization
3227CC	Fragmentation of forest; bush encroachment; loss of plantations; invasion
3228BA	Bush encroachment; invasion
3228BD	Loss of plantations; bush encroachment; fragmentation; some riparian invasion
3324CD	Plantations to cultivation; forest to plantation, shrubs and cultivation; invasion; fragmentation of forest; reduction in forest; plantations in former forest; loss of plantation; plantation to forest edge
3326DB	Bush encroachment; loss of plantations; fragmentation of forests; invasion

Below follow descriptions of the range declines of each species, as well as possible causal agents of these declines. This includes the results of the occupancy modelling, the changes in land cover concurrent with the species' loss from a site, and particulars about the species' habitat and ecology that may explain these changes in range.

Rufous-chested sparrowhawk (*Accipiter rufiventris,* Smith 1830)

This bird of prey had a range decline of 36% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to losses in plantations (Table 3.2). It was lost from 10/30 sites (Table 3.1), all of which suffered losses and fragmentation of plantations (Tables 3.3 and 3.4). This species is naturally found in indigenous montane forest (Sinclair *et al.* 2011; BirdLife International 2014a), and is thought to have spread into the south-western parts of the country by using alien plantations and alien invasion of exotic pines and eucalyptus for nesting (Macdonald 1989; Simmons 1986; Hockey and Midgley 2009).

African goshawk (*Accipiter tachiro,* Daudin 1800)

This bird of prey had a range decline of 5% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be correlated with increases in waterbodies, while the species increased with urbanization and its initial distribution was dependent on plantations (Table 3.2). It was lost from 17/30 sites (Table 3.1), most of which lost natural vegetation (largely indigenous forests), some of which lost plantations, and all of which had an increase in waterbodies (Tables 3.3 and 3.4). It occurs naturally in indigenous montane, lowland and mangrove forests, and is found in exotic plantations (BirdLife International 2014a) and suburban gardens (Sinclair *et al.* 2011). Many raptor species are known to drown in farm reservoirs, including several accipiters, although no records exist currently of the African goshawk drowning (Anderson *et al*. 1999; Anderson 2000).

Narina trogon (*Apaloderma narina*, Stephens 1815)

This insectivore had a range decline of 4% nationally. Although occupancy modelling did not predict this decline, its initial distribution was dependent on cultivation and plantations (Table 3.2). It was lost from 11/30 sites (Table 3.1), most of which had a loss of indigenous forest, and all of which had a

loss of plantations (Tables 3.3 and 3.4). It occurs naturally in indigenous dry, lowland and montane forest (BirdLife International 2014a), and is known to occur in plantations (BirdLife International 2014a). One study showed density of breeding pairs to decrease in forests utilized in fuel wood removal (Du Plessis 1995).

Lemon dove (*Aplopelia larvata,* Temminck 1809)

This omnivore had a range decline of 28% nationally. Although occupancy modelling did not predict this decline, it did indicate that the species initial range does not overlap with urbanized areas (Table 3.2). It was lost from 17/30 sites between SABAP1 and SABAP2 (Table 3.1). Fifteen of these sites experienced a loss of indigenous forest (Table 3.3). It is naturally found in indigenous lowland and montane forest (BirdLife International 2014a).

Cape batis (*Batis capensis,* Linnaeus 1766)

This insectivore had a range decline of 1% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to a loss of plantations (Table 3.2). It was lost from 11/30 sites (Table 3.1), nine of which lost indigenous forest, and nine of which lost plantations (Tables 3.3 and 3.4). Additionally, this species newly colonized one site between SABAP1 and SABAP2 (Table 3.1), with this site experiencing alien invasion and cultivated lands being allowed to return to their natural state over this time period (Tables 3.3 and 3.4). The Cape batis occurs naturally in indigenous lowland, montane and temperate forest (BirdLife International 2014a), and is known to occur in suburban gardens (Sinclair *et al.* 2011) and to forage in plantations (Earlé and Oatley 1983; Armstrong and van Hensbergen 1994).

Barratt's warbler (*Bradypterus barratti,* Sharpe 1876)

This insectivore had a range decline of 4% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to a loss of natural vegetation, and an increase in cultivation, urbanization, waterbodies and plantations (Table 3.2). It was lost from 10/30 sites (Table 3.1), all of which experienced deforestation of indigenous forests (Table 3.3). It newly colonized two

sites (Table 3.1), both with increased urbanization (Table 3.4). It is naturally found in indigenous montane forest (BirdLife International 2014a).

Forest buzzard (*Buteo trizonatus*, Rudebeck 1957)

This bird of prey had a range decline of 8% nationally. Although occupancy modelling did not predict this decline, its initial range was limited by urbanization and waterbodies (Table 3.2). It was lost from 10/30 sites (Table 3.1), seven of which had a loss of indigenous forests and all of which had an increase in waterbodies (Tables 3.3 and 3.4). It colonized two sites (Table 3.1), one of which had afforestation of grasslands with plantations, while the other experienced alien invasion of plantation edges, effectively increasing plantation size (Table 3.4). It is naturally found in indigenous montane forest (Sinclair *et al.* 2011), and utilizes plantations for range expansion, hunting and nesting (Tarboton 2001; Hockey and Midgley 2009; Sinclair *et al.* 2011). Farm reservoirs, the likely cause of the increase in waterbodies, are known to result in birds of prey drowning (Anderson *et al.* 1999; Anderson 2000).

Trumpeter hornbill (*Bycanistes bucinator,* Temminck 1824)

This omnivore had a range decline of 4% nationally. Occupancy modelling did not predict this decline. It was lost from 7/30 sites (Table 3.1), all of which suffered recent deforestation (Table 3.3). It is naturally found in indigenous dry, lowland and montane forest (BirdLife International 2014a), as well as suburban gardens (Sinclair *et al.* 2011). The trumpeter hornbill is known to be at risk from habitat loss, hunting and international trade (Trail 2007), and is utilized in the pet trade (BirdLife International 2014a). It feeds on exotic fruits and crops (Chittenden 2007), which could lead to persecution by farmers.

Green-backed camaroptera (*Camaroptera brachyura,* Vieillot 1820)

This insectivore had a range decline of 13% nationally. Although occupancy modelling did not predict this decline, its initial distribution was dependent on plantations and waterbodies (Table 3.2). It was lost from 8/30 sites (Table 3.1), six of which underwent deforestation of indigenous forests and four

of which lost plantations (Table 3.3). It occurs naturally in indigenous mangrove, lowland and montane forests, and occurs in plantations (BirdLife International 2014a).

Southern banded snake-eagle (*Circaetus fasciolatus,* Kaup 1850)

This raptor had a range decline of 16% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to a loss of cultivation and plantations, and its distribution was limited by urbanization (Table 3.2). It was lost from only one of the 30 sites studied (Table 3.1), which experienced a loss of indigenous forests (Table 3.3). It occurs naturally in indigenous lowland forest, and is known to occur in plantations (BirdLife International 2014a). It is globally near threatened, and regionally critically endangered (BirdLife South Africa 2014). It is at risk globally from biological resource use due to logging and wood harvesting, and is also utilized in the pet trade (BirdLife International 2014a).

Grey cuckooshrike (*Coracina caesia*, Lichtenstein 1823)

This insectivore had a range decline of 16% nationally. Occupancy modelling did not predict this decline. It was lost from 13/30 sites (Table 3.1), twelve of which experienced a loss of indigenous forest, and two of which experienced a loss of plantations (Table 3.3). It is naturally found in indigenous lowland and montane forests, and is known to occur in plantations (BirdLife International 2014a). It is at risk globally due to habitat destruction (del Hoyo *et al.* 2005).

Chorister robin-chat (*Cossypha dichroa,* Gmelin 1789)

This insectivore had a range decline of 20% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to an increase in waterbodies (Table 3.2). It was lost from 18/30 sites (Table 3.1), fifteen of which experienced a loss of indigenous forest, and all of which experienced an increase in waterbodies (Table 3.3). It is naturally found in indigenous lowland and montane forests (BirdLife International 2014a). The apparent sensitivity to increased water bodies is possibly as a result of the habitat loss and fragmentation caused by building dams, first suggested in a study by Wu *et al.* (2003) on the impacts of the Three Gorges Dam in China. Such impacts are not only apparent with large dams; Mantel *et al.* (2010) showed the effects of small dams to include

impacting water quantity and quality, as well as habitat structure (Stanley *et al.* 2002) and riparian vegetation (Jansson *et al.* 2000).

Forest canary (*Crithagra scotops,* Sundevall 1850) (previously *Serinus scotops*)

This seedeater had a range decline of 10% nationally. Occupancy modelling did not predict this decline. It was lost from 9/30 sites (Table 3.1), seven of which experienced a loss of indigenous forest, and three of which experienced a loss of plantations (Table 3.3). It is naturally found in indigenous lowland and montane forests (BirdLife International 2014a), as well as gardens (Hockey Midgley 2009) and plantation margins (Chittenden 2007).

Bush blackcap (*Lioptilus nigricapillus,* Vieillot 1818)

This frugivore had a range decline of 11% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to an increase in plantations, and was mitigated by an increase in cultivation and urbanization (Table 3.2). It was lost from 5/30 sites (Table 3.1), from which natural vegetation was lost in four (Table 3.3). The species also colonized one site (Table 3.1), where natural vegetation increased and forest expansion occurred (Tables 3.3 and 3.4). It is naturally found in indigenous montane forests (BirdLife International 2014a), and is known to occur in suburban gardens (Sinclair *et al.* 2011). It is globally near threatened, and regionally vulnerable (BirdLife South Africa 2014). This species is at risk globally from both wood and pulp plantations and agro-industry plantations (BirdLife International 2014a).

Mountain wagtail (*Motacilla clara,* Sharpe 1908)

This insectivore had a range decline of 30% nationally. Although occupancy modelling did not predict this decline, its distribution was found to be dependent on plantations (Table 3.2). It was lost from 12/30 sites (Table 3.1), nine of which experienced a loss of indigenous forest and four of which experienced a loss of plantations (Table 3.3). This species is largely restricted to forest surrounding rivers (Sinclair *et al.* 2011). The nature of plantation forestry legislation in South Africa limits plantations to be planted no closer than 10 m to existing streams; this allows the development of semi-natural riparian ecosystems (Malan *et al.* 2007) which may support species such as the mountain

wagtail. Similar strips of natural riparian vegetation found in pine plantations in Chile are thought to support forest bird populations (Estades and Temple 1999).

Southern double-collared sunbird (*Nectarinia chalybea,* Linnaeus 1766) (previously *Cinnyris chalybeus*)

This nectarivore had a range decline of 12% nationally. Although occupancy modelling did not predict this decline, its initial distribution was dependent on natural vegetation (Table 3.2). It was lost from 8/30 sites (Table 3.1), seven of which experienced deforestation of indigenous forests (Table 3.3). It is naturally found in indigenous lowland forest (BirdLife International 2014a), as well as scrub and fynbos (Sinclair *et al.* 2011).

Eurasian golden oriole (*Oriolus oriolus,* Linnaeus 1758)

This omnivore had a range decline of 35% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to increasing plantations, and is mitigated by increasing urbanization, while its initial distribution was limited by natural vegetation (Table 3.2). It was lost from 4/30 sites (Table 3.1), all of which experienced deforestation of indigenous forest, and two of which experienced an increase in plantations (Tables 3.3 and 3.4). This species is a Palearctic migrant (Sinclair *et al.* 2011). It is found naturally in dry, montane and (globally) temperate forest (BirdLife International 2014a), and occurs in exotic plantations and savannah (Sinclair *et al.* 2011), as well as suburban gardens (Patón 2013). It is utilized in the pet trade, for food and for sport (BirdLife International 2014a).

Yellow-throated woodland-warbler (*Phylloscopus ruficapilla,* Sundevall 1850)

This insectivore had a range decline of 21% nationally. Occupancy modelling did not predict this decline. It was lost from 11/30 sites (Table 3.1), ten of which experienced a loss of indigenous forest (Table 3.3). It increased in one site (Table 3.1), which experienced an increase in natural vegetation (Table 3.3). It is naturally found in indigenous dry, lowland and montane forests (BirdLife International 2014a).

White-starred robin (*Pogonocichla stellata,* Vieillot 1818)

This omnivore had a range decline of 23% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to an increase in plantations, and was mitigated by increased urbanization (Table 3.2). It was lost from 13/30 sites (Table 3.1), all of which experienced a loss of indigenous forest, and eleven of which experienced a loss of plantations (Table 3.3). It is naturally found in indigenous lowland and montane forests, and occurs in plantations (BirdLife International 2014a).

Cape parrot (*Poicephalus robustus,* Gmelin 1788)

This frugivore had a range decline of 58% nationally, the largest decline of the species studied. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to a loss of natural vegetation (Table 3.2). It was lost from 7/30 sites (Table 3.1), all of which experienced a loss of indigenous forest, and one of which experienced a loss of plantations (Table 3.3). It is found naturally in indigenous mangrove and montane forests, and is known to occur in plantations (BirdLife International 2014a). These parrots are food nomads (Wirminghaus *et al.* 2000), and feed in orchards (Wirminghaus *et al.* 2002; Sinclair *et al.* 2011) and plantations (Wirminghaus *et al.* 2002), as well as utilizing predominantly *Podocarpus* species for feeding (Wirminghaus *et al.* 2002) and nesting (Wirminghaus *et al.* 2001a). The conservation of *Podocarpus* forests is thought to be important for species persistence (Wirminghaus *et al.* 2000), and food availability is thought to be a reason for the species' decline (Wirminghaus *et al.* 2002). It is regionally endangered (BirdLife South Africa 2014), is utilized in the pet trade (BirdLife International 2014a) and is at risk from destruction by farmers when the birds raid orchards (Wirminghaus *et al.* 2002). A recent study by Downs *et al.* (2014) on the Cape Parrot's range over 15 years found an increase in species abundance in several sites, though the proportion of locations in which the species was recorded, decreased.

Buffspotted flufftail (*Sarothrura elegans,* Smith 1839)

This insectivore had a range decline of 29% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to a loss of plantations (Table 3.2). It had a range

decline nationally of 29%, and was lost from eleven sites (Table 3.1), all of which experienced a loss of indigenous forest, and eight of which experienced a loss of plantations (Table 3.3). It is naturally found in indigenous lowland, montane and swamp forest, and occurs in plantations (BirdLife International 2014a). It uses alien vegetation to increase its range (Hockey and Midgley 2009), and is found in suburban gardens (Sinclair *et al.* 2011).

African crowned eagle (*Stephanoaetus coronatus,* Linnaeus 1766)

This bird of prey had a range decline of 14% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to an increase in waterbodies (Table 3.2). It was lost from 18/30 sites (Table 3.1), sixteen of which experienced a loss of indigenous forest, and four of which experienced a loss of plantations (Table 3.3). Seventeen sites had an increase in waterbodies (Table 3.3). It increased in one site (Table 3.1), where there was mass scale invasion by exotic vegetation of natural, unwooded vegetation, and indigenous forest expansion (Table 3.4). It is naturally found in indigenous dry, lowland and montane forest, and occurs in plantations (BirdLife International 2014a), where it feeds on monkeys, small antelope and hyraxes (Sinclair *et al.* 2011). It is globally near threatened, and regionally vulnerable (BirdLife South Africa 2014). This species is at risk globally from a wide variety of threats, including agro-industry farming and plantations, shifting agriculture, small-holder farming, intentional and unintentional hunting and trapping, logging and wood harvesting and mining and quarrying. It is also utilized in the pet trade (BirdLife International 2014a). Additionally, many raptor species are known to drown in farm reservoirs, although no records exist currently of the African crowned eagle drowning (Anderson *et al.* 1999; Anderson 2000).

African wood owl (*Strix woodfordii,* Smith 1834)

This bird of prey had a range decline of 12% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to a loss of plantations (Table 3.2). It was lost from eleven sites (Table 3.1), ten of which experienced a loss of indigenous forest, and eight of which experienced a loss of plantations (Table 3.3). It is naturally found in indigenous lowland and montane forest (BirdLife International 2014a), and occurs in plantations (Sinclair *et al.* 2011).

Knysna turaco (*Tauraco corythaix,* Wagler 1827)

This omnivore had a range decline of 4% nationally. Although occupancy modelling did not predict this decline, its initial distribution was dependent on natural vegetation (Table 3.2). It was lost from 11/30 sites (Table 3.1), ten of which experienced a loss of indigenous forest (Table 3.3). It is naturally found in indigenous lowland and montane forest (BirdLife International 2014a).

Olive bush-shrike (*Telophorus olivaceus,* Shaw 1809)

This omnivore had a range decline of 2% nationally. Although occupancy modelling did not predict this decline, its initial distribution was dependent on plantations (Table 3.2). It was lost from 8/30 sites (Table 3.1), seven of which experienced a loss of indigenous forest, and two of which experienced a loss of plantations (Table 3.3). It is naturally found in indigenous dry, lowland and montane forests, and occurs in plantations (BirdLife International 2014a), as well as gardens (Hockey and Midgley 2009). It is at risk globally due to habitat destruction (Harris and Franklin 2000).

Blue-mantled crested flycatcher (*Trochocerus cyanomelas,* Vieillot 1818)

This insectivore had a range decline of 6% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to increases in plantation cover (Table 3.2). It was lost from 9/30 sites (Table 3.1), eight of which experienced a loss of indigenous forest and five of which experienced an increase in plantations (Table 3.3). It colonized one site (Table 3.1). It is naturally found in indigenous montane and lowland forests, and does not occur in plantations (BirdLife International 2014a).

Orange ground-thrush (*Zoothera gurneyi,* Hartlaub 1864)

This insectivore had a range decline of 8% nationally. Within the thirty sites studied here, occupancy modelling showed its decline to be attributed to increases in plantations and urbanization, and was mitigated by increases in cultivation, natural vegetation and waterbodies (Table 3.2). It was lost from 6/30 sites (Table 3.1), all of which experienced a loss of indigenous forest, and five of which experienced an increase in urbanization (Table 3.3). Plantations increased in five sites (Tables 3.3 and 3.4), and natural vegetation decreased in four sites (Table 3.3). It is naturally found in indigenous

montane forests, and does not occur in plantations (BirdLife International 2014a). It is regionally near threatened (BirdLife South Africa 2014).

Spotted ground-thrush (*Zoothera guttata,* Vigors 1831)

This insectivore had a range decline of 28% nationally. Occupancy modelling did not predict this decline. It was lost from two sites (Table 3.1), both of which experienced a loss of natural vegetation including a decrease in indigenous forest (Table 3.3). It is naturally found in indigenous dry, lowland and montane forests (BirdLife International 2014a). It is globally and regionally endangered (BirdLife South Africa 2014). This species is at risk globally from a wide variety of threats, including agro-industry farming, logging and wood harvesting (BirdLife International 2014a).

Discussion

The main cause of loss of the forest dependent species identified here is a loss of indigenous forest. This is aggravated in the case of 50% (n = 14) of these species by a loss of plantation forests, which have been shown in Mauritius (Carter and Bright 2002) and Malaysia (Mitra and Sheldon 1993) to act as a refuge for certain species tolerant of plantations, or a corridor between small forest fragments, allowing a rescue effect in the forests of KwaZulu-Natal, South Africa (Wethered and Lawes 2003).

Habitat loss is the biggest cause of species extinction globally (Pimm and Raven 2000), and loss of indigenous forest is the largest driver of local extinction in the forest dependent birds studied here (see species accounts in Results, this chapter, and Tables 3.1-3.4). After 1994, Participatory Forest Management in South Africa allowed increased access to and use of forest resources by rural populaces (Grundy and Michell 2004). Combined with a breakdown in traditional forest authorities, and a lack of governmental resources to monitor and police reserves (Lawes *et al.* 2004b), this has led to a large-scale degeneration and loss of indigenous forests (Grundy and Michell 2004). This study indicates that this has negatively affected forest birds, particularly in the former homeland areas of the Transkei and Ciskei, which are now mostly in the Eastern Cape province, and the East Griqualand area of KwaZulu-Natal. In the Eastern Cape slash and burn agriculture, as well as invasion of forests

for settlements resulting in deforestation and denudation of the understory, are common (Grundy and Michell 2004). A detailed study by Olivier *et al.* (2013) examined the possibility of an extinction debt in birds of the coastal forests of KwaZulu-Natal, and predicted that 14 of these forest dependent birds would likely go extinct in the future due to the effects of current fragmentation and habitat destruction. Similar trends could be occurring across the country. Our study shows mass fragmentation of indigenous forests in South Africa (Table 3.4), leading to isolation of small fragments and the species therein. This fragmentation disrupts ecosystem processes, and contributes to biodiversity and species loss (Berliner 2009).

There is some disagreement on the effects of plantations on biodiversity, with two conflicting views presented in the literature: that plantations improve biodiversity of adjacent indigenous forests (Estades and Temple 1999; Bremer and Farley 2010), or that plantations reduce biodiversity of adjacent indigenous forests (Geldenhuys 1991; Wethered and Lawes 2003, 2005), and alter species assemblages (Armstrong and van Hensburgen 1995; Allan *et al*. 1997). The results of this study show that of the eleven species affected by plantations, half are affected positively and half are affected negatively (Table 3.2). It has previously been postulated that certain guilds or groups of species exhibit the same reaction to plantations (e.g. Prestt 1965; Armstrong and van Hensburgen 1995); however, this was not evident in our study, with no particular guild appearing to benefit from or be impaired by plantations (Table 3.2). Plantations may act as a refuge for those species tolerant to them (e.g. Carter and Bright 2002), where indigenous forests are lost. Some species have developed such a tolerance that they prefer to breed and feed in plantations, such as many birds of prey (Tarboton 2001; BirdLife International 2014a). Some relative specialists, such as the Cape parrot, also feed in plantations (Wirminghaus *et al.* 2002). South Africa experienced an increase in plantations towards the end of the last century (Berliner 2009), before decreasing over the last twenty years (Table 3.3; Forestry Economics Services CC 2014). This decrease in plantation cover could be leading to a loss of species which feed and breed in plantations, as well as those which use plantations to buffer the effects of indigenous forest loss. In addition, the forests themselves may be buffered by plantations from local harvesting pressure (Berliner 2009) and loss to fire (Geldenhuys 2002). If a loss of

plantations leads to a loss of this protection of indigenous forests, even those species which do not occur in plantations may be negatively affected. Lantschner *et al.* (2008), in a study of birds in pine plantations in Argentina, suggested that species which evolved in a fragmented forest biome may be preadapted to surviving in plantations, as they have evolved to withstand some levels of disturbance. The long history of natural fragmentation of forests in South Africa (Berliner 2009) could enable South African forest birds to do the same. Losses in indigenous forest and plantations alike may be somewhat mitigated by urbanization and cultivation, where species can survive in gardens or cultivated land, if this urbanization and cultivation is not the cause of indigenous forest and plantation loss.

Birds of prey are known to drown in farm reservoirs and dams, although none of the species in this study were listed as having drowned in the studies of Anderson *et al.* (1999) or Anderson (2000). The increase in the percentage of the sites covered by waterbodies between 1994 and 2014 seen here (Table 3.3) is perhaps indicative of a loss of indigenous habitat associated with dam construction, although in most cases the percentage increase is small. Visual inspection reveals increased dam construction, probably mostly for irrigation, in both farm and communal land. Almost all species had increased waterbodies as one of the factors determining their initial range (Table 3.2); this probably simply reflects the fact that forests grow in areas which are well-resourced in terms of water and are therefore appropriate sites for dam construction.

Of the 28 species with declining ranges, 24 were secondary consumers (birds of prey, insectivores or omnivores which feed on insects or invertebrates). Higher trophic levels are more at risk from habitat destruction, alteration and fragmentation (Shoener 1968; Ewers and Didham 2006), leading to a trophic bias in response to human-mediated habitat loss (Duffy 2003). Hunting and harvesting of local resources is common in South African rural communities (Shackleton and Shackleton 2004), and species utilized often comprise birds (e.g. Shackleton *et al.* 2002; Twine *et al.* 2003; Shackleton and Shackleton 2006), including birds of prey (Asibey 1974). Other threats to birds of prey in areas utilized by humans include deliberate and accidental poisoning, gin traps, drowning in farm

reservoirs, electrocution by and collision with power lines, and road casualties (Anderson 2000). Insectivorous birds are known to decrease with increasing urbanization (Chace and Walsh 2006), and this decline is attributed to a loss of invertebrate food resources (Wilson et al. 1999; Benton et al. 2002).

There was no geographic pattern relating to characteristics of species lost, although most sites, and therefore species losses, occurred in the Eastern Cape province (see Addendum Figure A1 and Table 3.1). The sites with the greatest loss of species were 3129AB (18 species lost), and 3226BC (15 species lost) (Table 3.1), both of which are in the Eastern Cape province (Addendum Figure A1). Both of these sites experienced a loss of plantations and natural vegetation, an increase in cultivation, deforestation of indigenous forests (Table 3.3) and mass-scale invasion of alien trees (Table 3.4). Loss of plantations, combined with a loss of indigenous forests and degradation of remaining forests through their invasion by exotic species, could be impacting species diversity. Further natural habitat destruction be avoided, as should fragmentation or removal of plantations without mitigation such as restoration of indigenous forests.

Overall, the Eastern Cape contained more QDGC experiencing a loss of species (Addendum Figure A1). The sites in the Eastern Cape had a higher proportion of sites with a loss of plantations and an increase in cultivation than the sites in the other provinces (Table 3.3). This loss of plantations, combined with a loss of indigenous forests and degradation of remaining forests through their invasion by exotic species, could therefore be impacting species diversity. It is strongly encouraged that further habitat destruction be avoided, as well as fragmentation or removal of plantations, shown here to act as a refuge for certain forest bird species, without mitigation such as restoration of indigenous forests. The Eastern Cape has poor conservation implementation brought about by lack of manpower to actively police forest fragments and reserves (Lawes *et al.* 2004b). Local harvesting is poorly regulated, and is prominent in the rural parts of the Eastern Cape especially, with inadequate protection and unsustainable management (Grundy and Michell 2004). The results of this study

suggest that these forest fragments and reserves need to be better protected to prevent species loss. The Eastern Cape is destined to experience a further increase in plantations, as it is the only province in which surplus water is available, and the Department of Agriculture, Fisheries and Forestry is pursuing a target of 10 000 ha of nett new afforestation nationally a year. This policy should be pursued with extreme caution in the light of the findings of this study. Although plantations can benefit some forest-dependent bird species, each individual case for afforestation needs to be carefully assessed in terms of its potential impact on the matrix in which indigenous forest fragments are embedded.

Chapter 4 - A trait-based risk assessment of South African forest birds

Cooper, TJG[1]; Norris, K[2]; Wannenburgh, A[3]; Cherry, MI[1]
[1]Department of Botany and Zoology, Stellenbosch University; [2]Zoological Society of London;
[3]Department of Environmental Affairs, South Africa

Abstract

Forests in South Africa are under threat from a variety of threats. Among these are anthropogenic threats, including land use changes, resulting in the biota of this biome being at risk. However, the impacts of these changes on wildlife populations are poorly understood. A pan-European trait-based risk assessment framework was modified for use in South Africa. This framework allows the identification of habitats and species most at risk, as well as the most important threats to species persistence. The original model was modified through the addition of any positive effects of land use change on species, to create a net risk model. Montane forests were found to be more at risk than other forest types. The major risks facing these forests were increased abundance of small predators, increased fire suppression, increased soil management, removal of deadwood and reduced diversity of tree species. These threats are all products of plantation forestry and local harvesting. Nesting risk was found to be higher than foraging risk across all species. Montane forests should be better protected to preserve these species, and negative effects from plantation forestry and local harvesting should be mitigated. No differences were found in the results of the traditional and net models, suggesting that the negative impacts of land use change outweigh any benefits to bird species in South African forests.

Introduction

The forest biome in South Africa occupies only 0.56% of the country's land, and much of it has been extensively fragmented due to climate changes during the Quaternary and anthropogenic activities more recently (Low and Rebelo 1996; Eeley *et al.* 1999; Berliner 2009). Despite its small contribution to the country's landscape, the forest biome is home to a large proportion of the country's biodiversity, including 14% of the country's bird species (Geldenhuys and MacDevette 1989). In addition, forests comprise the highest percentage of threatened mammal, bird, amphibian and reptile taxa of any biome in South Africa (Berliner 2009). The greatest threats to the forest biome in South

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Africa currently are anthropogenic fragmentation (Berliner 2009; Lawes *et al.* 2006); deforestation for agriculture and plantations (Geldenhuys and MacDevette 1989); and degradation caused by rural harvesting (Cocks and Wiersum 2003; Shackleton and Shackleton 2004).

Bird species are a useful biodiversity monitoring tool within forests (Furness and Greenwood 1993) due to their sensitivity to ecosystem changes and the ease with which they are recognized, making monitoring programs by both scientists and the public feasible (Wade *et al.* 2013). Many citizen science projects aimed at monitoring species exist, and the South African Bird Atlas Project (SABAP) is one such project. Changes in range size for bird species between the first SABAP, run from 1987-1992, and the second SABAP, run from 2007 and on-going, allows the prospect of investigating changes in avian distribution over this time period.

Previous work on forest birds in South Africa has included studies on the biodiversity of avifauna in various forests in South Africa (Brown 2006; Oatley 1989; Cooper 1985; Cody 1983), studies on the effects of plantations on avifauna in South Africa (Symes *et al.* 2000; Allan *et al.* 1997; Armstrong and van Hensbergen 1995), studies on bird response to habitat loss (Grass *et al.* 2013), studies on the effects of habitat conversion (Péron and Altwegg 2015a; Loftie-Eaton 2014) and individual species studies (e.g. Downs *et al.* 2014; Wirminghaus *et al.* 2002; Simmons 1986; Earlé and Oatley 1983).

Species characteristics may influence response to habitat loss (Ewers and Didham 2006). Many studies have successfully used species' traits to determine their risk to habitat loss (e.g. Butler *et al.* 2007; Butler *et al.* 2009; Wade *et al.* 2013; Wade *et al.* 2014). Butler *et al.* (2007) developed a trait-based risk assessment framework to determine the influence of agriculture on farmland birds in the United Kingdom, which was later applied to European farmland (Butler *et al.* 2010) and forest birds (Wade *et al.* 2013). This framework uses a species' ecological requirements in terms of diet, foraging habitat and nesting habitat to assess the impact of land use change on population change. To do this, it produces a score that describes the risks that land use changes pose to a species' ecological requirements, with a larger score denoting greater risk. Previous work has shown that when applied to

historical land use changes, the risk scores correlate significantly with population changes, with populations with higher scores declining at a greater rate. It is thus possible to use the risk scores calculated in this framework to identify habitats and species experiencing the greatest risk from land use changes, as well as identify the specific land use changes associated with these risks. This assessment can then be used to guide conservation interventions. This framework can potentially be used prospectively, to determine future risks to species, but is here used retrospectively, to determine reasons for previous range change.

Recent research on forest dependent birds in South Africa has analyzed the changes in land use data to determine likely causes of range declines in these species over the last twenty years (Cooper *et al.* submitted). Here we aim to identify current risks to forest dependent bird species in South Africa, to predict which of these risks will most influence species in the future. The model of Wade *et al.* (2013) applied here additionally makes provision for separate risk prediction for foraging and nesting. In particular, we predict that the prevalence of fuel wood harvesting and local harvesting in South Africa (Cocks and Wiersum 2003; Shackleton and Shackleton 2004) could lead to species' nesting habitats being more at risk than their foraging habitats. In doing so, we present a new model which incorporates positive effects of land use changes on species into a 'net risk' model.

Methods

Species selection and forest type delineation

Bird species were identified as being forest dependent by consultation of Oatley's list of forest dependent birds (1989) in conjunction with BirdLife International's list of highly forest dependent species (BirdLife International2014a). A review of current literature allowed description of species-specific characteristics. Characteristics were split into foraging and nesting, and within each the type of forest utilized, horizontal habitat used (edge or core) and vertical habitat used (ground, understory or canopy) were delineated. Additionally, the food items utilized (below-ground invertebrates, above-ground invertebrates, seeds, plant-based foods and vertebrates) and the nest type used (hole in dead

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wood, hole in living wood, non-breeding migrant or external – any nest type not within a tree) were assigned. The original models used population estimates (Butler *et al.* 2007; Butler *et al*. 2009; Butler *et al.* 2010; Wade *et al.* 2013; Wade *et al.* 2014). However, the South African Bird Atlas Project (SABAP) does not calculate population estimates, but rather reporting rates. There are several problems associated with using SABAP reporting rates as a proxy for population estimates (pers. comm. Assoc. Prof. Res Altwegg), and so changes in range size, based on presence-absence data, were used to predict how species responded to various risks. Range sizes in 1992 and in 2014 were obtained from SABAP1 and 2. Range sizes for SABAP2 (which is ongoing) were taken as at 30 September 2014, and comprised the number of quarter-degree grid cells occupied by a species within South Africa.

South African forest categorization varies between three (Eeley *et al.* 2001), ten (Cooper 1985), twelve (Mucina and Rutherford 2006), fifteen (Acocks 1953) and 24 forest types (von Maltitz *et al.* 2003). As only 57 forest dependent bird species were analyzed here, use of any of the more recent forest categorizations reduced sampling sizes to unusable levels with no statistical significance to discern the risk to individual forest types. To provide meaningful results, a broad categorization system was used here, comprising montane, lowland and "other" forest types. The latter category was a catch-all category into which many of the smallest forest types were lumped. This category is the least meaningful of the three, but the small geographic range and selection of species within these individual forest types made any other categorization unfeasible. This categorization was chosen as it aligns with the classification of BirdLife International (2014a), the data on which species habitats were based, and so allowed continuity. These categories can be seen in Table 4.1.

Table 4.3 Categorization of forest types used in this study, and those by von Maltitz *et al.* (2003) and Mucina and Rutherford (2006).

This study	Von Maltitz *et al.* 2003	Mucina and Rutherford 2006
Montane	Marekele afromontane forest	Northern afrotemperate forest
	Northern highveld forest	Northern mistbelt forest
	Drakensberg montane forest	
	Northern KwaZulu-Natal mistbelt forest	
	Northern mistbelt	
	Mpumalanga mistbelt forest	
	Eastern mistbelt forest	
	Transkei mistbelt forest	
Lowland	Western Cape talus forest	Southern afrotemperate forest
	Western Cape afrotemperate forest	Southern mistbelt forest
	Southern Cape afrotemperate forest	Southern coastal forest
	Amatole mistbelt forest	Northern coastal forest
	Eastern scarp forest	Lowland riverine forest
	Pondoland scarp forest	Scarp forest
	Transkei coastal scarp forest	
	KwaZulu-Natal coastal forest	
Other	KwaZulu-Natal dune forest	Sand forest
	Eastern Cape dune forest	Ironwood dry forest
	Albany coastal forest	Swamp forest
	Western Cape milkwood forest	Mangrove forest
	Lowveld riverine forest	
	Swamp forest	
	Mangrove forest	
	Licuati sand forest	

Risk assessment

The risk assessment framework allows the comparison of the risk to a species from changes in land use and the species' range growth over the same period. This methodology was first published by Butler *et al.* (2007), with further applications by Butler *et al.* (2009), Butler *et al.* (2010), Wade *et al.* (2013) and Wade *et al.* (2014). Major changes to forest habitats were identified, and their impacts on forest resources assessed. For each forest bird species, use of these resources was detailed, and coincidence of resource use and impacts on resources allowed the calculation of a risk score for each species in the study. This risk score was adjusted for ecological resilience, defined as niche breadth and reliance on forest resources (Wade *et al.* 2013). General Linear Modelling was then used to determine the relationship between risk score and changes in range size, with the expectation that risk score would be negatively correlated with range change. Several adjustments to the original model had to be made, including re-assessment of the risks to determine applicability to South Africa, the use of range size data instead of population estimates, and the introduction of a net risk model, which had the addition of risks being allowed to have positive as well as negative impacts on populations in

certain circumstances, such as those species which forage and breed in plantations benefitting from the addition of plantations to an area.

Modelling risks

To validate the framework, changes in land use which impacted birds' use of foraging or nesting components were assessed. 18 risks were identified as being applicable within a South African context (see Addendum Table A3). This included 14 of the original risks identified by Wade *et al.* (2013), as well as four unique to the South African context, as identified according to the current literature (see Addendum Table A3 for details on this literature). Where applicable, relevant statistics and proxies were used to determine the effects of these risks. This methodology was followed by the authors of the framework, and was adapted for South Africa through the utilization of South African studies. See Butler *et al.* (2007), Wade *et al.* (2013) and Addendum Table A3 for more details.

The effects of each of the 18 risks on montane, lowland and other forest types, was determined according to the relevant literature (Geldenhuys 1989; von Maltitz *et al.* 2003; Berliner 2005), as positive (+), negative (-) or neutral (0). This is an extension of the work by Butler *et al.* (2007) and Wade *et al.* (2013), in the acceptance of positive effects of risks on species. The net effects of all risks to each forest type was then calculated by subtracting the total number of positive effects acting on a resource from the total number of negative effects, or risks, resulting in a net risk score which was used for further analyses. See Table S4 for this calculation. As only one country was studied, risk scaling per country was not necessary.

For each bird species, habitat use was identified as per the current literature, as the trait being affected within a habitat space (1), or not (0). Habitat spaces utilized included both those used in foraging and in nesting. In foraging, feeding on below-ground invertebrates, above-ground invertebrates, plant material, seeds and vertebrates was described. In nesting, nests in holes in dead wood, in holes in living wood, external nests and non-breeding migrants were described (Du Plessis 1995; Tarboton 2001; del Hoyo *et al.* 1994; del Hoyo *et al.* 1996; del Hoyo *et al.* 1997; del Hoyo *et al.* 1999; del

Hoyo *et al.* 2001; del Hoyo *et al.* 2002; del Hoyo *et al.* 2003; del Hoyo *et al.* 2004; del Hoyo *et al.* 2005; del Hoyo *et al.* 2006; del Hoyo *et al.* 2007; del Hoyo *et al.* 2008; del Hoyo *et al.* 2009; del Hoyo *et al.* 2010). For both foraging and nesting, vertical and horizontal habitat use were described (ground, understory and canopy, and edge or core, respectively) (Tarboton 2001; del Hoyo *et al.* 1994; del Hoyo *et al.* 1996; del Hoyo *et al.* 1997; del Hoyo *et al.* 1999; del Hoyo *et al.* 2001; del Hoyo *et al.* 2002; del Hoyo *et al.* 2003; del Hoyo *et al.* 2004; del Hoyo *et al.* 2005; del Hoyo *et al.* 2006; del Hoyo *et al.* 2007; del Hoyo *et al.* 2008; del Hoyo *et al.* 2009; Mirski 2009; del Hoyo *et al.* 2010), as was the use of plantations and alien trees for foraging or nesting (Tarboton 2001; del Hoyo *et al.* 1994; del Hoyo *et al.* 1996; del Hoyo *et al.* 1997; del Hoyo *et al.* 1999; del Hoyo *et al.* 2001; del Hoyo *et al.* 2002; del Hoyo *et al.* 2003; del Hoyo *et al.* 2004; del Hoyo *et al.* 2005; del Hoyo *et al.* 2006; del Hoyo *et al.* 2007; del Hoyo *et al.* 2008; del Hoyo *et al.* 2009; del Hoyo *et al.* 2010). The literature used was chosen because of the level of detail available with regard to horizontal and vertical habitat use.

Following this, the coincidence between habitat use and risks to habitat space was calculated for each species. This coincidence between habitat use and the risks to this habitat space was then used to calculate niche breadth scores as the proportion of habitat types used by a species. Risk scores for foraging and nesting were calculated, with foraging risk as below

$$D_t = \frac{A}{D * F} + B/F$$

And nesting risk as below

$$N_t = \frac{C_1}{N_1} + \frac{C_2}{N_2}$$

Where D_t: total diet risk; A: points of coincidence between risk impact and diet use; D: number of diet components used; F: number of foraging components used; B: points of coincidence between risk impact and foraging use; N_t: total nesting risk; C_1: points of coincidence between risk impact and nest

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type used; N_1: number of nest components used; C_2: points of coincidence between risk impact and nesting site used; and N_2: number of nesting sites used.

Total risk was calculated as the sum of diet and foraging risk. Total risk per species for each model was then modeled against changes in the range size of that species to determine possible reasons for bird range change, using Generalized Linear Models (GLMs) in R (family = Poisson) (R Core Team 2014). Total risk was disaggregated into nesting and foraging risk, to determine which of these uses was most impacted by threats to forests, as well as into the initial 18 risks (Addendum Table A3), to determine specific threats to species, by allowing each risk to have a different weighting (Wade *et al.* 2013). Nesting and foraging risks were modeled against changes in range size using GLMs, as was each individual risk.

A net risk model was then run. This net risk model furthered the traditional model through the addition of positive effects of land use changes on species. The effects of each of the 18 risks were categorized as positive, negative or neutral. The net effects of all risks to each forest type was then calculated by subtracting the total number of positive effects acting on a resource from the total number of negative effects, resulting in a net risk score (see Addendum Table A4). This net risk score was modeled against changes in range size using GLMs.

Correlations were done between traditional and net model risk scores to determine how well the net risk model functioned, and t-tests were done to determine difference in fit between traditional and net models. To analyze why risks to montane forests were found to be significant and risks to lowland forests were not, when the risks faced by the two forest types and the risks scores of the species found in each forest type were so similar, the risks scores of the species occurring in each forest type were correlated to determine which forest type had a higher risk score overall. This was done using species which occurred in only montane or lowland forest; those which occurred in both were excluded from this analysis. To determine which forest type was most at risk, average risk scores for each forest type were compared, and a one-way ANOVA was done to determine differences in foraging risk and

nesting risk among montane, lowland and other forest types. R was used for all analyses. An outlying species, the migratory European honey buzzard (*Pernis apivorus*) was removed from modeling as the only species to have an increase in range of more than 100%, or eight times that of the species with the next greatest change in range size.

Results

Traditional measure

The highest total risk score combining all forest types, summed, was 10, for both the spotted ground-thrush (*Zoothera guttata*) and the crowned hornbill (*Tockus alboterminatus*). These were closely followed by the trumpeter hornbill (*Bycanistes bucinator*), the green barbet (*Stactolaema olivacea*) and the yellow-throated woodland warbler (*Phylloscopus ruficpilla*) with 9.75, and the narina trogon (*Apaloderma narina*) and the African crowned eagle (S*tephanoaetus coronatus*) with 9.6. Within montane forests, the highest risk score was 3.5, for the Knysna turaco (*Tauraco corythaix*), the blue-mantled crested-flycatcher (*Trochocerus cyanomelas*) and the orange ground-thrush (*Zoothera gurneyi*). Within lowland forests, the highest risk score was 3.5, for the Knysna turaco and the blue-mantled crested-flycatcher. Within other forest types, the highest risk score was 3.3, for the African crowned hornbill and the spotted ground-thrush. Average nesting risk was higher than average foraging risk for all forests (3.754 and 2.437 respectively), as well as for montane forests (1.263 and 0.855), lowland forests (1.579 and 0.979) and other forests (0.912 and 0.603). Average niche breadth (risk score/reliance) was higher for lowland forests (1.562), than for montane (1.39) and other forests (0.988). Risk scores for all species can be seen in Addendum Table A5.

The best fitting models were for the montane forests, and all of these were significantly negative (Table 4.2). Total risk to montane forest was negatively correlated with change in range size (p = 0.00382), as was foraging risk (p = 0.00604) and nesting risk (p = 0.00438). Change in range size was not significantly related to risk score for either lowland forest or other forest types. Average change in bird species' range size within each forest type was -4.49% for montane forest (n = 39), 7.08% for

lowland forest (n = 45) and 5.23% for other forest types (n = 28), indicating that species ranges are declining in montane forests, whereas in lowland and other forest types species ranges are mostly increasing. Nesting risk was significantly higher for montane forests than lowland or other forests (p < 0.05), as was foraging risk (p < 0.001).

Table 4.2 Results of the GLM for the traditional method.

Model	AIC	Estimate	p	F
Growth ~ All-Total	546,2064	-2,037	0,212	1,594
Growth ~ All-Foraging	545,8904	-5,694	0,173	1,908
Growth ~ All-Breeding	546,4677	-2,987	0,253	1,335
Growth ~ Montane-Total	539,0804	-9,697	0,00382	9,138
Growth ~ Montane-Foraging	539,9461	-22,657	0,00604	8,169
Growth ~ Montane-Breeding	539,3397	-15,426	0,00438	8,846
Growth ~ Lowland-Total	547,4573	2,319	0,548	0,3656
Growth ~ Lowland-Foraging	547,4502	5,8686	0,544	0,3725
Growth ~ Lowland-Breeding	547,4643	3,81	0,552	0,3588
Growth ~ Other-Total	547,6736	-1,408	0,694	0,1561
Growth ~ Other-Foraging	547,3755	-6,005	0,507	0,4451
Growth ~ Other-Breeding	547,788	-1,231	0,832	0,04555

Net measure

Net risk scores for all forest types combined were highest for the crowned hornbill and the spotted ground-thrush (10), and the trumpeter hornbill and the green barbet (9.75). Net risk scores for montane forests were highest for the Knysna turaco, the blue-mantled crested-flycatcher and the spotted ground-thrush (3.5). Net risk scores for lowland forests were highest for the Knysna turaco and the blue-mantled crested-flycatcher (3.5). Net risk scores for other forests were highest for the crowned hornbill and spotted ground-thrush (3.3). Average risk scores were higher for nesting risk than foraging risk across all forest types (3.386 and 1.996 respectively), as well as in montane (1.137 and 0.695 respectively), lowland (1.42 and 0.805 respectively) and other forests (0.829 and 0.496 respectively). Niche breadth was higher in lowland forests (1.387) than in montane (1.207) and other forests (0.873). Risk scores for all species can be seen in Addendum Table A6.

The best fitting models were for the montane forests, which had a significantly negative relationship between change in range size and risk score (Table 4.3; total risk p = 0.0119, foraging risk p = 0.0355, nesting risk p = 0.00848). Change in range size was not significantly related to risk in either lowland forest or other forest types.

Table 4.3 Results of the GLM for the net method.

Model	AIC	Estimate	p	F
Growth ~ All-Total	546,8666	-1,656	0,336	0,9422
Growth ~ All-Foraging	546,9033	-4,057	0,345	0,9061
Growth ~ All-Breeding	546,8988	-2,636	0,344	0,9105
Growth ~ Montane-Total	541,2214	-9,248	0,0119	6,77
Growth ~ Montane-Foraging	543,2086	-18,982	0,0355	4,651
Growth ~ Montane-Breeding	540,5862	-15,439	0,00848	7,466
Growth ~ Lowland-Total	547,1959	3,1087	0,435	0,62
Growth ~ Lowland-Foraging	547,2117	7,6561	0,44	0,6045
Growth ~ Lowland-Breeding	547,1969	5,1393	0,435	0,619
Growth ~ Other-Total	547,6591	-1,599	0,682	0,1701
Growth ~ Other-Foraging	547,2816	-7,36	0,467	0,5365
Growth ~ Other-Breeding	547,7896	-1,297	0,835	0,044

Disaggregation by risk

When split by risk type, the reasons for the discrepancy in fit among forest types becomes clear (see Addendum Table A7). For four risks, the relationship between range change and the risk score of all forest types is significantly negative ($p < 0.05$). However, for three of these only montane forests are affected. One risk was significantly negative only for montane forests only, while one risk was significantly positive for montane forests. Risk from increased abundance of small predators is significantly negatively correlated to range change in montane forests ($p = 0.00858$, Figure 4.1), as is risk from increased fire suppression ($p = 0.0189$, Figure 4.2), intensified soil management ($p = 0.0128$, Figure 4.3), removal of deadwood ($p = 0.0298$, Figure 4.4), and reduced diversity of tree species ($p = 0.0189$, Figure 4.5). Increased grazing pressure was positively correlated with species range growth in montane forests ($p = 0.0456$, Figure 4.6).

Increased abundance of small predators
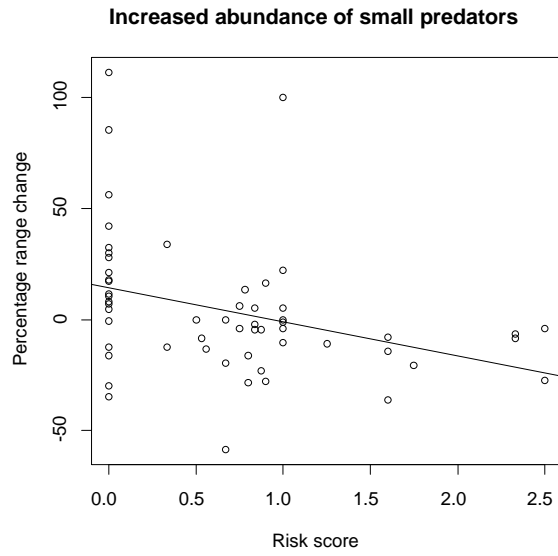
Figure 4.1 The relationship between risk of increased abundance of predators and range change in montane forests.

Increased fire suppression
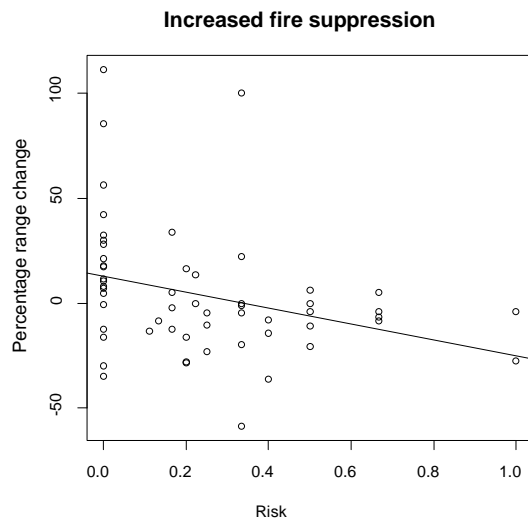
Figure 4.2 The relationship between risk of increased fire suppression and range change in montane forests.

Intensified soil management
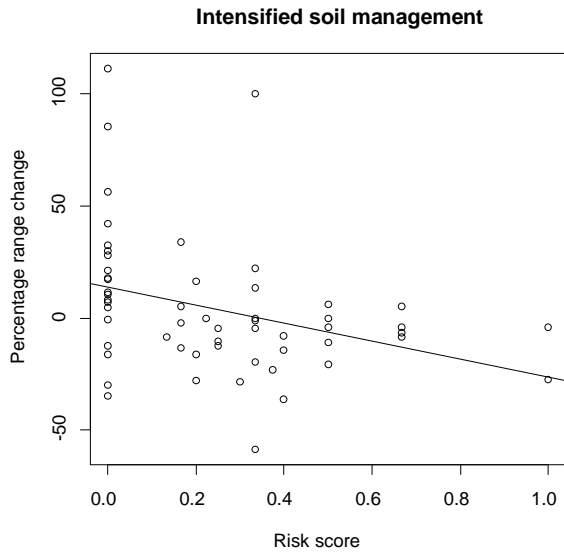
Figure 4.3 The relationship between risk of intensified soil management and range change in montane forests.

Removal of deadwood
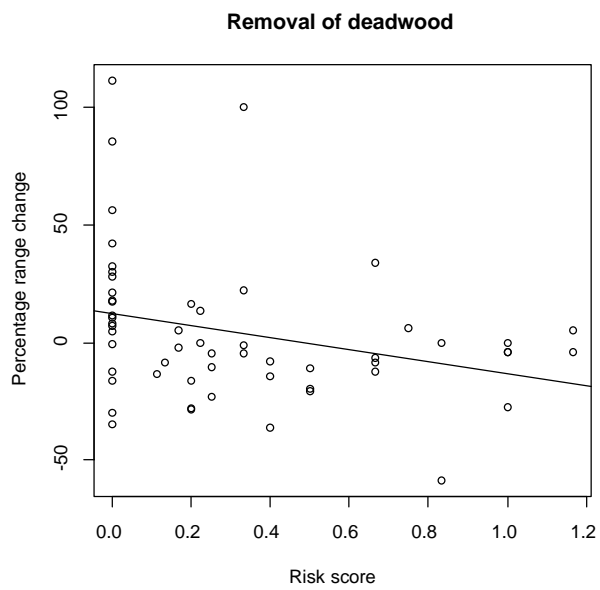
Figure 4.4 The relationship between risk of deadwood removal and range change in montane forests.

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Decreased diversity of tree species
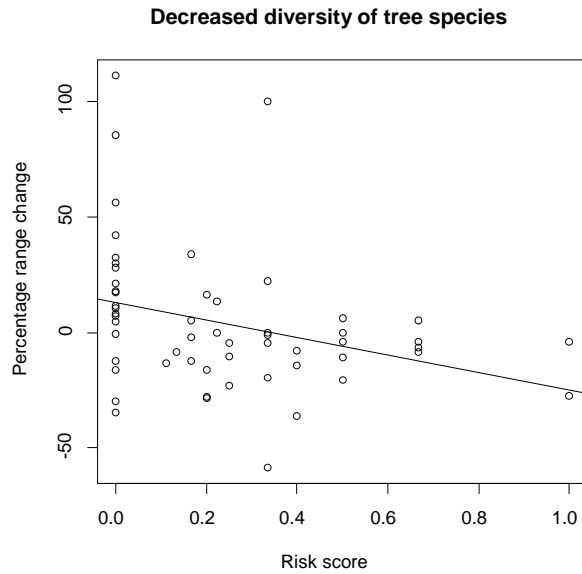
Figure 4.5 The relationship between risk of decreased tree species diversity and range change in montane forests.

Increased grazing pressure
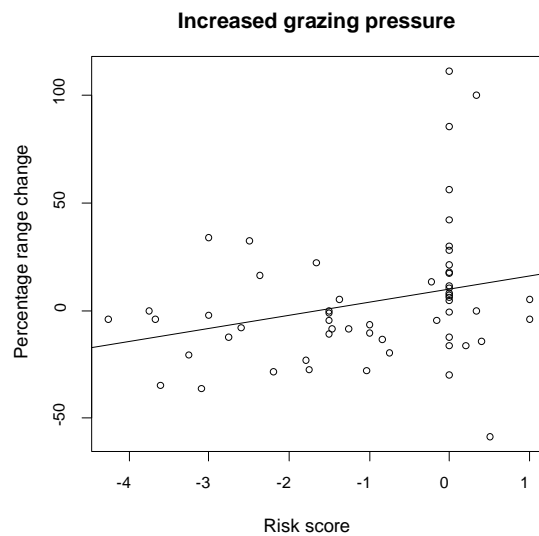
Figure 4.6 The relationship between risk of increased grazing pressure and range change in montane forests.

Montane vs. Lowland species

The risk scores of species occurring only in montane or lowland forests were compared to determine why only risk to montane forest was significant. It was predicted that the risk scores of species occurring only in montane forests would be higher, and this would result in the difference in GLM fit. Results of these comparisons showed no significant relationships between change in species' range size and risk scores for either species occurring only in montane forests, or species occurring only in lowland forests. The effect size for total risk within all forests was -13.737 for montane-only species,

and 12.724 for lowland-only species. The effect size for montane-only species within montane forests was -0.4196 for total risk, -6.289 for foraging risk and 2.206 for nesting risk, while the effect size for lowland-only species within lowland forests was 2.724 for total risk, 7.892 for foraging risk and 3.926 for nesting risk. Additionally, the correlation between average risk scores of montane-only and lowland-only species was 0.905877. However, risk scores for species occurring only in montane forests were almost all negatively correlated with change in range size, while risk scores of species occurring only in lowland forests were all positively correlated with change in range size. Accordingly, it seems as though risks are having a more negative effect on montane forests than lowland forests. See Table 4.4 for the list of montane, lowland and montane and lowland species.

Table 4.4 Bird species found only in montane forest, only in lowland forest or in both montane and lowland forest, used to compare the risks to these forest types.

Montane only	Lowland only	Montane and lowland
Rufous-chested sparrowhawk	Barthroated apalis	African goshawk
Accipiter rufiventris	*Apalis thoracica*	*Accipiter*
Barratt's warbler	Woodward's batis	Narina trogon
Bradypterus barratti	*Batis fratrum*	*Apaloderma narina*
Forest buzzard	Yellowbellied greenbul	Lemon dove
Buteo trizonatus	*Chlorocichla flaviventris*	*Aplopelia larvata*
Pinkthroated twinspot	African emerald cuckoo	Cape batis
Hypargos margaritatus	*Chrysococcyx cupreus*	*Batis capensis*
Bush blackcap	Southern banded snake-eagle	Trumpeter hornbill
Lioptilus nigricapillus	*Circaetus fasciolatus*	*Bycanistes bucinator*
Eurasian golden oriole	Olive sunbird	Green-backed camaroptera
Oriolus oriolus	*Cyanomitra olivacea*	*Camaroptera brachyura*
Cape parrot	Square-tailed drongo	Eastern bronze-naped pigeon
Poicephalus robustus	*Dicrurus ludwigii*	*Columba delegorguei*
Orange ground-thrush	Bearded scrub-robin	Grey cuckooshrike
Zoothera gurneyi	*Erythropygia quadrivirgata*	*Coracina caesia*
	Grey waxbill	Chorister robin-chat
	Estrilda perreini	*Cossypha dichroa*
	Crested guineafowl	Forest canary
	Guttera edouardi	*Crithagra scotops*
	Scalythroated honeyguide	Olive woodpecker
	Indicator variegatus	*Dendropicos griseocephalus*
	Black-bellied starling	Brown scrub-robin
	Lamprotornis corruscus	*Erythropygia signata*
	Green-backed twinspot	Collared sunbird
	Mandingoa nitidula	*Hedydipna collaris*
	Southern double-collared sunbird	Yellow-streaked greenbul
	Nectarinia chalybea	*Phyllastrephus flavostriatus*
	Dark-backed weaver	Yellow-throated woodland-warbler
	Ploceus bicolor	*Phylloscopus ruficapilla*
	Livingstone's turaco	Yellow-rumped tinkerbird
	Tauraco livingstonii	*Pogoniulus bilineatus*
		White-starred robin
		Pogonocichla stellata
		Buffspotted flufftail
		Sarothrura elegans
		African broadbill
		Smithornis capensis
		Green barbet

	Stactolaema olivacea
	African crowned eagle
	Stephanoaetus coronatus
	African wood-owl
	Strix woodfordii
	Knysna turaco
	Tauraco corythaix
	Black-fronted bush-shrike
	Telophorus nigrifrons
	Olive bush-shrike
	Telophorus olivaceus
	Crowned hornbill
	Tockus alboterminatus
	Blue-mantled crested-flycatcher
	Trochocerus cyanomelas
	Tambourine dove
	Turtur tympanistria
	Spotted ground-thrush
	Zoothera guttata

Comparing traditional and net risk

Although the traditional and net methods give different risk scores, these are strongly correlated (Table 4.5; Figure 4.7), with a t-test between the AIC results of the traditional and net models showing very strong similarity between the two sets of models indicating that the net risk model explains results as well as the traditional model does (t = -0.0045; DF = 118; p = 0.9964).

Table 4.5 The correlation of net and traditional models, showing high correlation for each forest type and both foraging and nesting risk.

Data	Coefficient
All forest, total risk	0,973582
All forest, foraging risk	0,954971
All forest, nesting risk	0,981726
Montane forest, total risk	0,971914
Montane forest, foraging risk	0,929627
Montane forest, nesting risk	0,98706
Lowland forest, total risk	0,981213
Lowland forest, foraging risk	0,917001
Lowland forest, nesting risk	0,985042
Other forest, total risk	0,987493
Other forest, foraging risk	0,979002
Other forest, nesting risk	0,990839
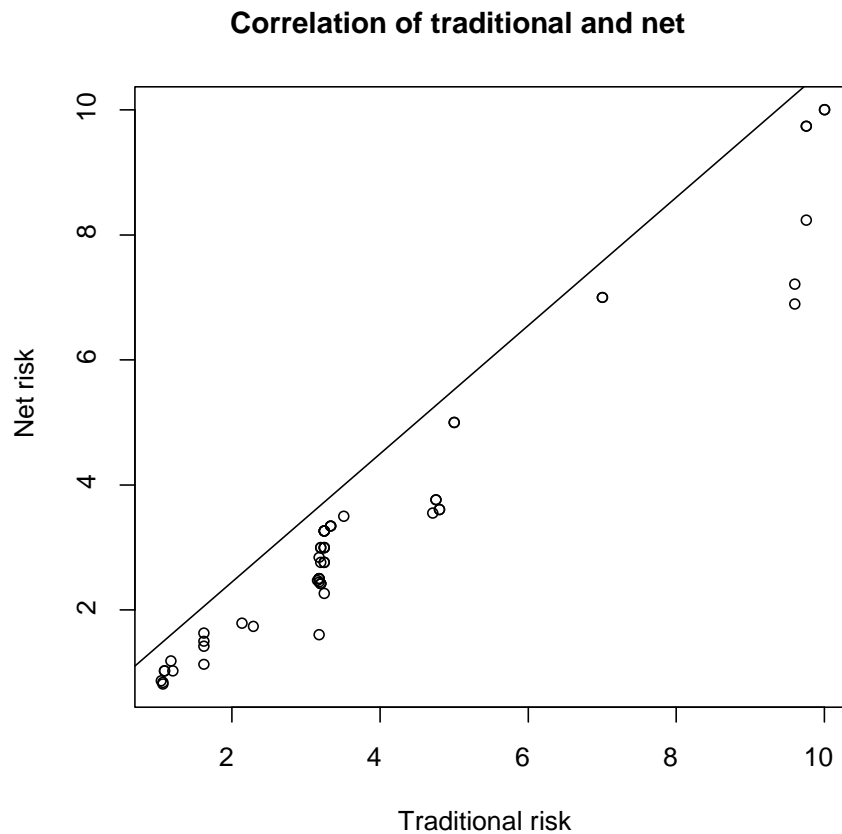
Figure 4.7 The correlation of traditional and net model risk scores, for all forests and total risk.

Discussion

Risks to forest dependent birds in South Africa

Unlike European studies (Wade *et al.* 2013), nesting risk was calculated to be higher than foraging risk for South African species (see Addendum Table A5). Some species of forest birds in South Africa are known to forage in plantations and farmlands surrounding indigenous forests (Earlé and Oatley 1983; Armstrong and van Hensbergen 1994; Wirminghaus *et al.* 2001a; Downs 2005; Hockey & Midgley 2009; BirdLife International 2014c), perhaps mitigating the effects of risks to forest habitat on foraging. Local harvesting of fuel wood and plant material is common in South Africa (Cocks and Wiersum 2003; Shackleton and Shackleton 2004), limiting the materials available for nesting. In addition, plantations limit nesting opportunities both for hole-nesting insectivores and species which build nests in forest undergrowth (Armstrong and van Hensburgen 1995; Estades and Temple 1999). Accordingly, the individual risks to South African forests have a higher impact on nesting due to a

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limitation in nesting material and suitable nesting sites, and so the reproductive success of forest dependent bird species will be impacted as their nesting resources are most at risk.

The main threats found to South African forest birds were in montane forests, a system not previously thought to be at high risk (Eeley *et al.* 2001). The risks to this forest type were (1) increased abundance of small predators, generally resulting in reduced nest success (Brawn *et al.* 2001). Plantations attract rodents normally not found in South African indigenous forests (Willan 1984); these species may feed on the eggs of bird species nesting in the plantations. (2) Increased fire suppression, generally resulting in a reduction in shrub and early- to mid-succession nesting sites (Esseen *et al.* 1997; Brawn *et al.* 2001; Betts *et al.* 2010). Fire suppression is known to occur in indigenous forests adjacent to plantations in South Africa (Berliner 2009). (3) Intensified soil management associated with plantation forestry, generally reducing ground nesting (de Jong *et al.* 2008) and foraging opportunities (Esseen *et al.* 1997). (4) Removal of deadwood, used by some hole-nesting birds (Robles *et al.* 2011); local harvesting of fuel wood is common in South African forests (Berliner 2009). (5) Reduced diversity of tree species as a result of selective local harvesting and plantations, generally resulting in a reduction in food resources (Brockerhoff *et al.* 2008; Jactel *et al.* 2009). In addition, reduced diversity of tree species may impact nesting, with previous studies in Canada having found that forest stands with multiple tree species support more shrub-, canopy- and cavity-nesting birds than forest stands with only one tree species (Hobson and Bayne 2000).

Increased grazing pressure was found to improve range growth in forest dependent birds within montane forests. Grazing has two main effects pertinent to this study: at an individual level, grazing causes plants to grow "bushier", and at a community level, can slow the succession process; both of these effects provide habitats for species that forage and nest in the undergrowth (Ryder 1980). However, the same study found that some forest and woodland birds in the United States are negatively impacted in the long term by grazing, as the understory will eventually be depleted through grazing (Ryder 1980), while other species appear to benefit from grazed forests (Ryder 1980; Donald *et al.* 1998). This difference appears to be caused by foraging height preference, as well as level of

grazing, with moderate levels found to increase plant structural diversity and to be preferred by some bird species (Martin and Possingham 2005).

All of the threats significantly affecting forest dependent birds in South Africa are by-products of plantation forestry and local harvesting in indigenous forests (see Addendum Table A3 for sources), which are known to be some of the major threats to South African forests (Geldenhuys and MacDevette 1989; Cocks and Wiersum 2003; Shackleton and Shackleton 2004). This study shows which impacts of these activities most endanger forest bird species. Appropriate mitigation strategies can thus be implemented in montane forests in particular to ensure that these threats do not cause further loss of species. Such strategies could include the implementation of more environmentally-friendly plantation forestry practices. Paquette and Messier (2010) suggest that plantation forestry can be beneficial to biodiversity, by retaining some live trees during harvest, preparing the site before harvest to limit soil disturbance, allowing the growth of an understory, and allowing greater age of plantations. Local harvesting impacts deadwood abundance in natural forests, however if the scale of this harvesting is limited within natural forests where possible, such as in protected reserves, the effects on hole-nesting species can be limited. However, there is some issue with implementation, as the montane forests which are most at risk here are less protected and managed, having lost their national protection in 1994 and experienced a demise resulting from a lack of protection and sustainable management since (Grundy and Michell 2004). In addition, this could be why only montane forests had a significant negative relationship between risk scores and species' range change. Lowland forests of South Africa are often associated and interspersed with plantations, and so are protected by the plantation authorities (pers. comm. J. Feely). In the coastal rural areas of the Eastern Cape, (which comprise predominantly lowland forests in this study), per annum consumption of fuel wood is lower than that of southern KwaZulu-Natal (which comprises predominantly montane forests in this study), suggesting that less deadwood may be removed from lowland forests (Lawes *et al.* 2004b). However, as fuelwood consumption in the indigenous forests of the Eastern Cape has been shown to approximate deadwood production by the indigenous forests themselves (Obiri 2002), the utilization in montane forests must then exceed sustainable levels.

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The species with the highest risk scores in this study were the spotted ground-thrush (*Zoothera guttata*), the crowned hornbill (*Tockus alboterminatus*), the trumpeter hornbill (*Bycanistes bucinator*), the green barbet (*Stactolaema olivacea*), the yellow-throated woodland-warbler (*Phylloscopus ruficpilla*), the narina trogon (*Apaloderma narina*), the African crowned eagle (*Stephanoaetus coronatus*), the Knsyna turaco (*Tauraco corythaix*), the blue-mantled crested-flycatcher (*Trochocerus cyanomelas*) and the orange ground thrush (*Zoothera gurneyi*). Four of these species are regionally threatened: the green barbet is known to occur in only one forest in South Africa (Sinclair *et al.* 2011) and is regionally endangered; the African crowned eagle is regionally vulnerable and globally near threatened; the orange ground-thrush is regionally near threatened; and the spotted ground-thrush is both regionally and globally endangered (BirdLife South Africa 2014).

The high risk scores of the six species which do not appear on the IUCN Red List give new insight into which species may be at risk regionally, and may soon be threatened. Both hornbill species and the narina trogon all make use of cavity nests (Tarboton 2001), and logging of older trees and removal of dead wood would limit available nesting options for these species. The Knysna turaco, yellow-throated woodland warbler and blue-mantled crested-flycatcher all occur in the interior of forests and have decreasing ranges (Sinclair *et al.* 2011; SABAP data, this study). Fragmentation and loss of forest habitats would restrict the suitable habitat of these species. To prevent these species from becoming threatened, environmentally-friendly plantation practices are again recommended, as is the prevention of further fragmentation of indigenous South African forests.

An interesting aspect of this study is that the Cape parrot (*Poicephalus robustus*) did not emerge as having a particularly high risk score, despite suffering the largest range loss of the species in this study, according to the SABAP data. This species is largely dependent on the yellowwood tree (*Podocarpus* spp.) for feeding and reproduction (Wirminghaus *et al.* 2001b; Downs 2005), and the coarse scale of this study did not allow quantification of the risks to these individual tree species, perhaps explaining why the Cape parrot was not determined as having high risk. Additional risks to the Cape parrot include the pet trade and being shot by farmers when feeding in orchards

(Wirminghaus *et al.* 1999). Both these risks are species-specific, and were not quantified in this study. Thus a limitation of this study is that risks which are species-specific and are not quantified as national risks could lead to certain species seeming to be less at risk than they are in reality. A study by Downs *et al.* (2014) examining the range of the Cape parrot over 15 years found while had its range declined, the abundance of the species had increased in the areas in which it remained. A confounding effect in relation to this particular species is the fact that it migrates locally in response to food resource availability (Wirminghaus *et al.* 2002) and so the time of year that individual sites were visited in the SABAP project may have influenced results. However, this was equally likely in both SABAP1 and SABAP2, and so the impacts of this are likely to be negligible.

Previous studies on forest birds in Europe found migration strategy and forest type to be the most reliable factors in determining risk to these species (Wade *et al.* 2013). Migration strategy was not included in this study, as only four species were migrants. Forest type did influence risk, with those species occurring in montane forest having higher risk scores than those occurring in lowland or other forest types, potentially because of the lack of protection in montane forests compared with lowland forests (Grundy and Michell 2004; Lawes *et al.* 2004b), as discussed previously.

Comparing net and traditional models

The strong correlation between the results of the net and traditional model (see Tables 4.2, 4.3, 4.5) shows that, in South Africa, there is no benefit to the addition of a net risk model, as any positive effects of land use change are outweighed enough by the negative effects as to not change the model outcome. This implies that risks are more significant than benefits within a South African context. This was unexpected, as those species which are capable of traversing plantations and making use of alien vegetation were predicted to benefit from these changes in land use. While many of the species in this study utilize plantations or forest edges for foraging or nesting (Tarboton 2001; Sinclair *et al.* 2011; BirdLife International 2014a), this study suggests that these adaptations are not sufficient to entirely protect these species from the negative effects of land use change.

Where the results of the models did differ was in the species found to be most at risk, with the traditional model resulting in four additional species having high risk scores. These were the yellow-throated woodland warbler, the narina trogon, and the crowned eagle. This is thought to be because these species utilize plantations or alien vegetation for foraging (del Hoyo *et al.* 1994; del Hoyo *et al.* 2001; del Hoyo *et al.* 2006; BirdLife International 2014a) or nesting (Tarboton 2001), and so allowing the positive impacts of this utilization in the calculation of the net risk score allowed these species to have a slightly lower net risk score than traditional risk score.

Conclusions

This model builds on previous models in the inclusion of positive effects of land use changes on species in a net risk model. However, despite some species having adapted to plantations and alien vegetation, risks outweigh benefits to the extent that the results of the net and traditional models do not differ.

The analysis of the SABAP1 and 2 data clearly shows that the ranges of more than half of South African forest dependent birds are declining (Cooper *et al.* submitted). The application of this model to these data gives insight into why this is occurring, beyond simple land use change. A collection of anthropogenic threats to species are narrowed down to several severe threats, indicating which land use practices in particular are most important in causing range declines in forest dependent birds, and which species are likely to suffer most in the near future if these detrimental practices are not mitigated. This study offers predictive results, which can potentially be used to ensure that further species are not placed on the IUCN Red List, or even become locally extinct.

Montane forests are most at risk, and so should benefit from conservation priority. The greatest threats to this forest type are a result of plantation forestry and some aspects of local harvesting. These threats should be alleviated through the proper implementation of existing legislation which limits of local harvesting within indigenous forests. However, such implementation is impeded by a lack of resources and manpower of the Department of Agriculture, Fisheries and Forestry, which limits its

ability to monitor and police its reserves, or community- or privately-owned forests (Lawes *et al.* 2004c; von Maltitz and Shackleton 2004).

Chapter 5 - Conclusions

Forests globally are at risk from a variety of anthropogenic threats, including deforestation, forest degradation, fragmentation, over-harvesting, invasion and climate change (Gardner et al. 2009; Tabarelli et al. 2004; Atkinson 1977) impacting the biodiversity within them. More than 50% of forests globally are compromised (FAO 2009). African forests suffered some of the highest levels of deforestation, second only to South America (FAO and JRC 2012). Threats to African forests specifically include fragmentation, subsistence hunting, and shifting cultivation (Barnes and Lahm 1997), leading to a decline in fauna and flora (e.g. Cordeiro and Howe 2003, 2001; Beier et al. 2002).

In South Africa, forests are naturally fragmented (Low and Rebelo 1996) but have suffered further extensive fragmentation and deforestation at the hands of humans (Lawes et al. 2006; Berliner 2009). The effects of anthropogenic fragmentation are thought to be amplified by the presence of exotic plantations in the matrix surrounding indigenous forest (Berliner 2009). These plantations cover double the area of indigenous forest in South Africa (Forestry Economics Services CC 2014; Low and Rebelo 1996). Plantations benefit indigenous forests by acting as corridors between forest patches (Wethered and Lawes 2003, 2005), acting as nurseries for indigenous trees (Geldenhuys 1997), and acting as a refuge for some species. However, plantations can cause a loss of biodiversity in indigenous forests (Geldenhuys 1991), alter species assemblages (Allan et al. 1997), and facilitate alien invasion (Geldenhuys 1997). The impacts of these threats to forests may differ with species (Neushulz et al. 2011; Ewers and Didham 2006).

This study aimed to ascertain the impact of changes in land cover on forest dependent bird ranges, through the use of SABAP data and land cover data. It was predicted that land use changes which negatively impacted the distribution of indigenous forest, for example increased urbanization or cultivation of natural vegetation, would lead to a reduction in forest dependent birds from the area. As the literature on the effects of plantations on forest dependent birds in South Africa is so discordant, it was predicted that the response of birds to plantations would differ at the species level. It was also

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predicted that fuel wood harvesting and local collection would make species nesting habitats more vulnerable than their foraging habitats, as several of the species studied here are known to forage outside of plantations (see Chapter 3 for details).

Occupancy was modelled to determine the relationship between changes in bird range size and changes in land use, focusing primarily on how land use changes have modified the habitat of forest dependent bird species. Of the 57 South African forest dependent birds identified here, 28 had decreasing ranges, demonstrating that changes in land use occurring in South Africa are largely detrimental to avian biodiversity of indigenous forests. As the species assessed here are all forest dependent bird species, these changes are likely to be impacting indigenous forest in South Africa. There were 30 QDGC identified as most at risk, with more than ten forest dependent bird species lost between SABAP1 and SABAP2. The majority of these were in the Eastern Cape, indicating that this province in particular is suffering forest degradation and/or deforestation. The forests in this province, and particularly those which formed part of the former 'homelands', have experienced many changes in their managing authorities (von Maltitz and Shackleton 2004). This, in conjunction with a lack of manpower and resources to police forest fragments and reserves and regulate local harvesting, has led to poor conservation implementation in the Eastern Cape (Grundy and Michell 2004; Lawes *et al.* 2004b). The Eastern Cape data were examined regionally to determine if this trend was simply a result of fewer report cards being submitted for the province in SABAP2; it was found that in the former Transkei and East Griqualand report numbers had increased from SABAP1 to SABAP2 in 67% of sites. The former Ciskei had decreased numbers of report cards for all sites; it is possible that for this region the loss of forest birds seen in the SABAP data a reflection of poorer sampling in this area.

The factors found to most impact the occupancy of forest dependent were changes in plantation cover and increased urbanization and cultivation, while natural vegetation was found to be replaced by cultivation and cultivated areas were found to be replaced by urbanization. A loss of natural vegetation led to a loss of bird species.

When species were examined individually (Chapter 3), using species characteristics and habitat preferences along with occupancy modelling to gain insight into range declines, it was found that the main cause of species loss appears to be a loss of indigenous forest. This is aggravated in some species by a loss of plantations, as these species may utilize plantations as a replacement habitat where indigenous forest is degraded or lost. Plantations are harmful for species intolerant of them, creating impassable barriers in the landscape. However, more than half of forest dependent bird species studied here are tolerant of plantations, and for these species plantations act as a refuge when indigenous forests are lost (e.g. Carter & Bright 2002), and as a corridor between fragments of indigenous forest (Wethered and Lawes 2003, 2005). The loss of indigenous forests due to exploitation, leading to fragmentation and habitat destruction, can be compounded by a loss of plantations. The species utilizing plantations in the face of indigenous forest loss are further affected by a subsequent loss of plantations. The species lost here are primarily secondary consumers, resulting from a trophic bias in the response of species to human-mediated habitat loss.

Healthy plant populations are maintained by pest-controlling insectivores; one of the most important of these has been shown in Europe to be cavity nesters (Bereczki *et al.* 2014). These are at risk in exploited forests, and forest biodiversity is under threat from anthropogenic land use change and deforestation. However, plantations, urban gardens and cultivated land may act as a biodiversity refuge for certain species. Plantations may mitigate local extinction in species which breed or feed in plantations. Increased urbanization has led to habitat loss, but urban gardens, if well-wooded, can support some forest dependent species, especially frugivores (Biggs *et al.* 2006). Previous studies globally have found that forest- and woodland species forage in cultivated land adjacent to their native habitats (Wilson *et al.* 1999; Hinsley and Bellamy 2000; Hughes *et al.* 2002) and certain South African species, such as the Cape parrot (Wirminghaus *et al.* 2001a; Downs 2005), are known to frequent orchards, suggesting that forest dependent species may utilize farmlands to supplement their diets.

Although no specific patterns were found in species loss across sites, sites in the Eastern Cape suffered more widespread plantation loss and more land transformation to cultivation than the other provinces studied, with 13/17 sites losing plantations and 10/17 sites increasing in cultivation (see Chapter 3, Table 12). This, as well as the larger number of at-risk sites in the Eastern Cape (see Chapter 1 Figure 1, and Addendum Figure 1A), indicates that these forests are particularly at risk from land use change, fragmentation and forest habitat loss. More research into the proper management of these forests is needed. Table 2.2 shows the disparity of report cards available for the Eastern Cape sites: the former Ciskei had a decline in number of report cards between SABAP1 and SABAP2 in all sites, while in the former Transkei and former East Griqualand this was only true for one third of sites. It is thus possible that the decline in report cards for the former Ciskei is the reason for the perceived species loss occurring here, rather than actual declines in species numbers. More studies are needed to confirm this. This study covers the species of only one taxonomic group. It is likely that similar trends of range decline are occurring in other forest dependent faunal groups. As bird species are arguably the most mobile taxonomic group it is likely that other groups are more negatively affected, unless bird species are specifically being targeted by humans. More study is needed to ascertain the trends of other taxonomic groups.

In the fourth chapter, a risk assessment framework was used to identify specific risks associated with land use change, as well as species' susceptibility to risk, and forest types most at risk. Montane forests are home to the majority of the 28 species with decreasing ranges, indicating that species within this forest type, and this forest type itself, are most at risk. The risks to this forest type in South Africa are largely products of plantation forestry and local harvesting, and include increased abundance of small predators, leading to increased nest predation (Brawn *et al.* 2001), increased fire suppression, leading to a reduction in understory vegetation and reducing nesting and foraging habitat in this layer (Esseen *et al.* 1997; Brawn *et al.* 2001; Betts *et al.* 2010), intensified soil management, negatively impacting ground-nesting (de Jong *et al.* 2008) and ground-foraging species (Esseen *et al.* 1997), removal of deadwood, limiting nesting habitats for cavity-nesting species (Robles *et al.* 2011),

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and decreased diversity of tree species, leading to homogenization of the habitat and a reduction of food (Brockerhoff *et al.* 2008; Jactel *et al.* 2009) and nesting (Hobson and Bayne 2000) resources.

Destruction of nesting habitat seems to be a larger risk to forest dependent birds than destruction of foraging habitat. This could be because species are utilizing plantations and farmlands for foraging, while local harvesting largely impacts potential nest sites. It was found that cavity nesters, as well as those which inhabit the interior of forest patches, may soon be at risk as a result of habitat loss and fragmentation.

The differences in the approaches of the three data chapters are as follows. Chapter 2 uses occupancy modeling to provide the most likely relationships between a change in bird range size and a given historic land use change, to identify which overarching changes in land use have impacted bird species. Chapter 3 goes further into detail on the species characteristics which may be making them susceptible to land use change, as well as the habitat requirements which are no longer met in certain sites, leading to local extinction from these sites. Chapter 4 uses risk modeling to determine how the ecological space that a species uses is impacted by the risks imposed by land use changes. This chapter also uses changes in bird range size and historic data on the changes in land use producing these risks, but gives further detail than Chapter 2 on which specific impacts of land use change have been influencing species range changes.

This study used data from both the South African Bird Atlas Projects (SABAP), run from 1987-1992 and 2007-present (although data used here was from 2007-September 2014), and land cover data from the National Land Cover Dataset 1994 (Thompson 1999), the National Land Cover 2000 (Van den Berg *et al.* 2008), the National Land Cover 2009 (for KwaZulu-Natal, Mpumalanga and Limpopo provinces) (SANBI 2009), the Eastern Cape Land Cover 2014 (for the Eastern Cape province) (CD: NGI 2010-2012), the South African National Land Cover Database (2013/2014) (GeoterraImage 2014). The changes witnessed in the SABAP data and the land cover data were not ground-truthed with site visits. It is suggested that further studies to corroborate these results be undertaken, by

visiting the thirty sites identified as most at-risk to assess land cover and bird species presence, and testing for the risks identified in Chapter 4, through comparisons of pristine and degraded forest.

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Addendum

Table A1 Forest dependent birds of South Africa, as per Oatley (1989) and BirdLife International (2014c).

Scientific name	Common name
Accipiter rufiventris	Rufous-chested sparrowhawk
Accipiter tachiro	African goshawk
Apalis thoracica	Barthroated apalis
Apaloderma narina	Narina trogon
Aplopelia larvata	Lemon dove
Aquila pomarina	Lesser spotted eagle
Batis capensis	Cape batis
Batis fratrum	Woodward's batis
Bradypterus barratti	Barratt's warbler
Buteo trizonatus	Forest buzzard
Bycanistes bucinator	Trumpeter hornbill
Camaroptera brachyura	Green-backed camaroptera
Chlorocichla flaviventris	Yellowbellied greenbul
Chrysococcyx cupreus	African emerald cuckoo
Circaetus fasciolatus	Southern banded snake-eagle
Columba delegorguei	Eastern bronze-naped pigeon
Coracina caesia	Grey cuckooshrike
Cossypha dichroa	Chorister robin-chat
Crithagra scotops	Forest canary
Cyanomitra olivacea	Olive sunbird
Dendropicos griseocephalus	Olive woodpecker
Dicrurus ludwigii	Square-tailed drongo
Erythropygia quadrivirgata	Bearded scrub-robin
Erythropygia signata	Brown scrub-robin
Estrilda perreini	Grey waxbill
Guttera edouardi	Crested guineafowl
Hedydipna collaris	Collared sunbird
Hypargos margaritatus	Pinkthroated twinspot
Indicator variegatus	Scalythroated honeyguide
Lamprotornis corruscus	Black-bellied starling
Lioptilus nigricapillus	Bush blackcap
Mandingoa nitidula	Green-backed twinspot
Motacilla clara	Mountain wagtail
Nectarinia chalybea	Southern double-collared sunbird
Oriolus oriolus	Eurasian golden oriole
Pernis apivorus	European honey-buzzard
Phyllastrephus flavostriatus	Yellow-streaked greenbul
Phylloscopus ruficapilla	Yellow-throated woodland-warbler
Ploceus bicolor	Dark-backed weaver
Pogoniulus bilineatus	Yellow-rumped tinkerbird
Pogonocichla stellata	White-starred robin
Poicephalus robustus	Cape parrot
Prionops retzii	Retz's helmet-shrike
Sarothrura elegans	Buffspotted flufftail
Smithornis capensis	African broadbill
Stactolaema olivacea	Green barbet
Stephanoaetus coronatus	African crowned eagle
Strix woodfordii	African wood-owl
Tauraco corythaix	Knysna turaco
Tauraco livingstonii	Livingstone's turaco
Telophorus nigrifrons	Black-fronted bush-shrike
Telophorus olivaceus	Olive bush-shrike
Tockus alboterminatus	Crowned hornbill
Trochocerus cyanomelas	Blue-mantled crested-flycatcher
Turtur tympanistria	Tambourine dove
Zoothera gurneyi	Orange ground-thrush
Zoothera guttata	Spotted ground-thrush

Table A2 The percentage of each response group to exhibit each characteristic studied.

Characteristic	Increasers	Decreasers	Stable
Occurs in plantations	45,45	53,57	14,29
Threatened	9,09	21,43	28,57
Montane	31,82	89,29	71,43
Mangrove	9,09	10,71	0,00
Lowland	81,82	71,43	100,00
Dry	54,55	25,00	57,14
Riverine	0,00	3,57	0,00
Swamp	4,55	3,57	0,00
Generalist	59,09	35,71	42,86
Specialist	40,91	64,29	57,14
Sedentary	81,82	67,86	85,71
Mobile	18,18	32,14	14,29
Resident	90,91	96,43	85,71
Migrant	9,09	3,57	14,29
Endemic	9,09	28,57	0,00
Not endemic	86,36	75,00	100,00
Edge	18,18	3,57	0,00
Interior	40,91	39,29	85,71
Edge and interior	40,91	57,14	14,29
Predator (mammals, birds, reptiles)	9,09	21,43	0,00
Omnivore	50,00	21,43	14,29
Frugivore	9,09	7,14	14,29
Insectivore	22,73	42,86	71,43
Seedeater	9,09	3,57	0,00
Nectivore	0,00	3,57	0,00
Small body size (<20cm)	68,18	50,00	85,71
Medium body size (20-50cm)	18,18	39,29	0,00
Large body size (50-80cm)	13,64	10,71	0,00
Very large body size (>80cm)	0,00	3,57	0,00
Built nest	59,09	64,29	57,14
Platform nest	0,00	17,86	0,00
Cavity nest	22,73	14,29	28,57
Scrape nest	4,55	0,00	0,00
Brood parasite	4,55	0,00	14,29
Non breeding migrant	9,09	3,57	0,00
Monogamous	86,36	96,43	85,71
Polygamous	4,55	0,00	14,29
Non-breeding migrant	9,09	3,57	0,00
Solitary nest dispersion	90,91	100,00	85,71
Social nest dispersion	4,55	0,00	0,00
Brood parasite	4,55	0,00	14,29
Nest site fidelity	22,73	50,00	28,57
Nest site infidelity	77,27	50,00	71,43
Low mass (<100g)	72,73	64,29	100,00
Medium mass (100-500g)	13,64	21,43	0,00
High mass (500-1000g)	4,55	10,71	0,00
Very high mass (>1000g)	9,09	3,57	0,00
Altricial	95,45	96,43	100,00
Precocial	4,55	3,57	0,00
1 egg	4,55	3,57	14,29
2 eggs	22,73	53,57	42,86
3 eggs	45,45	25,00	28,57
3+ eggs	27,27	17,86	14,29
<4 months breeding season	4,55	7,14	28,57
4-6 months breeding season	50,00	53,57	57,14
7-9 months breeding season	36,36	17,86	14,29
All year breeding season	9,09	21,43	0,00
Summer breeding	86,36	71,43	100,00
Winter breeding	0,00	7,14	0,00
All year breeding	13,64	21,43	0,00
Solitary	40,91	75,00	71,43
Gregarious	59,09	25,00	28,57

Table A3 Risks to forests in South Africa and their key impacts on forest birds. Sources are indicated by superscript numerals and are detailed below the table.

Change to forest habitat	Key impacts	Proxy (Wade et al. 2013) (if applicable)
1. Increased abundance of small predators	Reduced nest success of non-cavity nesters[6] Increased vertebrate prey*	Change in timber yield[35, 36]
2. Increased fire suppression[1, 2]	Reduction in invertebrate prey[7, 8] Reduction in shrub foraging habitat[6] Reduction in early and mid-succession foraging habitat[9] Reduction in shrub nesting sites[6,10] Reduction in early and mid-succession nesting habitat[9,10] Reduction in cavity nesting sites[7,10]	
3. Increased grazing pressure from domestic and wild herbivores[1]	Reduction in shrub foraging habitat[11,12] Reduction in quality of ground foraging habitat[11,12] Reduction in shrub and ground nesting sites[11,12] Reduction in nest success of ground nesters[11]	Change in cattle numbers[37]
4. Intensified drainage management	Reduction in below ground and ground dwelling invertebrate prey[13,14] Reduction in shrub foraging sites[15,16] Reduction in shrub nesting sites[15,16]	Change in timber yield[35, 36]
5. Intensified soil management	Reduction in below ground and ground dwelling invertebrates in early and mid-succession habitat[10,17] Reduction in quality of ground nesting sites in early and mid-succession habitat[17]	Change in timber yield[35, 36]
6. Intensified thinning	Reduction in shrub foraging habitat[18] Reduction in shrub nesting habitat[19]	Change in timber yield[35, 36]
7. Reduced abundance of canopy species	Reduction in canopy and shrub food resources (invertebrates/seeds/plant material)[20,21] Reduction in shrub and canopy nesting sites[22,23]	Change in timber yield[35, 36]
8. Reduced rotation length (including fragmentation effects)	Reduction in old growth foraging habitat[19,24] Reduction in core foraging habitat[25] Reduction in old growth succession nesting habitat[19,24] Reduction in core nesting habitat [25] Reduction in nesting success in edge habitat[6,26]	Change in timber yield[35, 36]
9. Removal of deadwood	Reduction in invertebrate prey[10] Reduction in cavity nest sites[27]	Change in timber yield[35, 36]
10. Reduced area of forest[3,4]	Reduction in broadleaf and mixed forest foraging and nesting habitat*	Change in area of forest[3]
11. Reduced diversity of tree species	Reduction in food resources (invertebrates/seeds/plant material)[21,23]	Change in timber yield[35, 36]
12. Increased forest fires[1]	Reduction in foraging and nesting habitat[28]	
13. Loss of habitat through urbanisation[1,3,4]	Reduction in foraging and nesting habitat*	
14. Increased selective logging	Reduction in invertebrates in closed canopy and old growth habitat[29] Reduction in cavity nests in closed canopy and old growth habitat[29]	Change in timber yield[35, 36]

15. Increased plantations[1,3,4]	Benefits species that forage and breed in plantations* Compacts soil, reducing below-ground inverts[30] Seed competition[30] Reduced edge habitat for foraging and nesting[30] Reduced shrub and canopy habitat for foraging and nesting[30]	Change in timber yield[35, 36]
16. Increased invasion by alien trees[1,5]	Benefits species that forage and breed in alien trees* Reduced above ground inverts, plants and seeds[31] Reduced edge and core habitat, shrub and canopy habitat[31] Benefits hole nesters, poor for external nesters[31]	
17. Increased fragmentation[1,3,4]	Reduced food resources and nesting resources* Increased edge habitat, reduced core habitat[32,33] Decreased undergrowth and canopy nesting and foraging habitat*	
18. Local harvesting[1]	Reduced food resources* Reduced edge and core habitat[34] Reduced vertical habitat[34]	

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*By definition

Table A4 Framework for the calculation of the net risk score for the net risk model.

	Foraging												Nesting										
	Diet					Foraging		Horizontal habitat		Vertical habitat			Nest type				Nesting		Horizontal habitat		Vertical habitat		
	Below-ground inverts	Above-ground inverts	Plant material	Seeds	Vertebrates	Plantation	Alien trees	Edge	Core	Ground	Shrub	Canopy	Hole - Dead wood	Hole - live wood	External	Non-breeding migrant	Plantation	Alien trees	Edge	Core	Ground	Understory	Canopy
Positive effects	0	0	0	0	1	1	1	1	0	0	0	0	1	2	0	0	1	1	1	0	0	0	0
Negative effects	6	9	7	7	2	1	1	6	8	6	12	9	7	7	7	0	1	1	7	8	8	13	10
Net effect	6	9	7	7	1	0	0	5	8	6	12	9	6	5	7	0	0	0	6	8	8	13	10
1. Increased abundance of small predators	0	0	0	0	1	0	0	0	0	0	0	0	0	0	-1	0	0	0	0	0	-1	-1	-1
2. Increased fire suppression	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
3. Increased grazing pressure from domestic and wild herbivores	0	0	-1	0	0	0	0	0	0	-1	-1	0	0	0	0	0	0	0	0	0	-1	-1	0
4. Intensified drainage management	-1	-1	0	0	0	0	0	0	0	0	-1	0	0	0	0	0	0	0	0	0	0	-1	0
5. Intensified soil management	-1	-1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	-1	0	0
6. Intensified thinning	0	0	0	0	0	0	0	0	0	0	-1	0	0	0	0	0	0	0	0	0	0	-1	0
7. Reduced abundance of canopy species	0	-1	-1	-1	0	0	0	0	0	0	-1	-1	0	0	0	0	0	0	0	0	0	-1	-1
8. Reduced rotation length (including fragmentation effects)	0	0	-1	-1	0	0	0	0	-1	0	-1	-1	-1	-1	0	0	0	0	-1	-1	0	-1	-1
9. Removal of deadwood	-1	-1	0	0	0	0	0	0	0	0	0	0	-1	0	0	0	0	0	0	0	0	0	0
10. Reduced area of forest	0	0	0	0	0	0	0	-1	-1	-1	-1	-1	-1	-1	-1	0	0	0	-1	-1	-1	-1	-1
11. Reduced diversity of tree species	0	-1	-1	-1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
12. Increased forest fires	0	0	0	0	0	0	0	-1	-1	-1	-1	-1	-1	-1	-1	0	0	0	-1	-1	-1	-1	-1
13. Loss of habitat through urbanisation	0	0	0	0	0	0	0	-1	-1	-1	-1	-1	-1	-1	-1	0	0	0	-1	-1	-1	-1	-1
14. Increased selective logging	0	-1	0	0	0	-1	-1	0	-1	0	0	-1	0	-1	0	0	-1	-1	0	-1	0	0	-1
15. Increased plantations	-1	0	0	-1	0	1	0	-1	0	-1	-1	0	0	1	0	0	1	0	-1	0	-1	-1	0
16. Increased invasion by alien trees	0	-1	-1	-1	0	0	1	-1	-1	0	-1	-1	1	1	-1	0	0	1	-1	-1	0	-1	-1
17. Increased fragmentation	-1	-1	-1	-1	-1	0	0	1	-1	0	-1	-1	-1	-1	-1	0	0	0	1	-1	0	-1	-1
18. Local harvesting (muthi harvesting)	-1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1	-1	-1	-1	0	0	0	-1	-1	-1	-1	-1

Table A5 Risk scores of forest dependent bird species describing their susceptibility to changes in land use, calculated the traditional way using only risks to species. Reliance describes how reliant a species is on forest ecosystems, with 1 implying a low reliance and 3 a high reliance. (AT) all forests, total risk, (AF) all forests, foraging risk, (AN) all forests, nesting risk, (MT) montane forests, total risk, (LT) lowland forests, total risk, (OT) other forest types, total risk, (MF) montane forests, foraging risk, (LF) lowland forests, foraging risk, (OF) other forest types, foraging risk, (MN) montane forests, nesting risk, (LN) lowland forest, nesting risk and (ON) other forest types, nesting risk.

Taxonomic name	Reliance	AT	AF	AB	MT	LT	OT	MF	MN	LF	LN	OF	ON
Rufous-chested sparrowhawk	1	3,142857	1,142857	2	3,142857	0	0	1,142857	2	0	0	0	0
African goshawk	2	4,714286	1,714286	3	1,571429	1,571429	1,571429	0,571429	1	0,571429	1	0,571429	1
Barthroated apalis	2	1,625	0,625	1	1,625	0	0	0	0	0,625	1	0	0
Narina trogon	1	9,6	3,6	6	3,2	3,2	3,2	1,2	2	1,2	2	1,2	2
Lemon dove	2	3,2	1,2	2	1,6	1,6	0	0,6	1	0,6	1	0	0
Lesser spotted eagle	1	1,2	1,2	0	1,2	0	0	1,2	0	0	0	0	0
Cape batis	2	3,25	1,25	2	1,625	1,625	0	0,625	1	0,625	1	0	0
Woodward's batis	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
Barratt's warbler	3	1,083333	0,416667	0,666667	1,083333	0	0	0,416667	0,666667	0	0	0	0
Forest buzzard	1	3,166667	1,166667	2	3,166667	0	0	1,166667	2	0	0	0	0
Trumpeter hornbill	1	9,75	3,75	6	3,25	3,25	3,25	1,25	2	1,25	2	1,25	2
Green-backed camaroptera	3	3,166667	1,166667	2	1,055556	1,055556	1,055556	0,388889	0,666667	0,388889	0,666667	0,388889	0,666667
Yellowbellied greenbul	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
African emerald cuckoo	2	1,625	0,625	1	0	1,625	0	0	0	0,625	1	0	0
Southern banded snake-eagle	1	3,2	1,2	2	0	3,2	0	0	0	1,2	2	0	0
Eastern bronze-naped pigeon	2	3,333333	1,333333	2	1,666667	1,666667	0	0,666667	1	0,666667	1	0	0
Grey cuckooshrike	2	3,2	1,2	2	1,6	1,6	0	0,6	1	0,6	1	0	0
Chorister robin-chat	2	3,2	1,2	2	1,6	1,6	0	0,6	1	0,6	1	0	0
Forest canary	2	3,2	1,2	2	1,6	1,6	0	0,6	1	0,6	1	0	0
Olive sunbird	2	3,166667	1,166667	2	0	1,583333	1,583333	0	0	0,583333	1	0,583333	1
Olive woodpecker	2	5	2	3	1,666667	1,666667	1,666667	0,666667	1	0,666667	1	0,666667	1
Square-tailed drongo	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
Bearded scrub-robin	2	3,166667	1,166667	2	0	1,583333	1,583333	0	0	0,583333	1	0,583333	1
Brown scrub-robin	2	5	2	3	1,666667	1,666667	1,666667	0,666667	1	0,666667	1	0,666667	1
Grey waxbill	3	1,083333	0,416667	0,666667	0	1,083333	0	0	0	0,416667	0,666667	0	0
Crested guineafowl	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
Collared sunbird	2	4,8	1,8	3	1,6	1,6	1,6	0,6	1	0,6	1	0,6	1
Pinkthroated twinspot	3	1,166667	0,5	0,666667	1,166667	0	0	0,5	0,666667	0	0	0	0
Scalythroated honeyguide	3	1,055556	0,388889	0,666667	0	1,055556	0	0	0	0,388889	0,666667	0	0
Black-bellied starling	1	3,333333	1,333333	2	0	3,333333	0	0	0	1,333333	2	0	0
Bush blackcap	2	1,625	0,625	1	1,625	0	0	0,625	1	0	0	0	0
Green-backed twinspot	3	1,047619	0,380952	0,666667	0	1,047619	0	0	0	0,380952	0,666667	0	0
Mountain wagtail	1	3,25	1,25	2	0	0	3,25	0	0	0	0	1,25	2
Southern double-collared sunbird	3	1,055556	0,388889	0,666667	0	1,055556	0	0	0	0,388889	0,666667	0	0
Eurasian golden oriole	1	2,285714	2,285714	0	1,142857	0	1,142857	1,142857	0	0	0	1,142857	0
Yellow-streaked greenbul	3	2,133333	0,8	1,333333	1,066667	1,066667	0	0,4	0,666667	0,4	0,666667	0	0
Yellow-throated woodland-warbler	1	9,75	3,75	6	3,25	3,25	3,25	1,25	2	1,25	2	1,25	2
Dark-backed weaver	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
Yellow-rumped tinkerbird	2	4,75	1,75	3	1,583333	1,583333	1,583333	0,583333	1	0,583333	1	0,583333	1
White-starred robin	2	3,25	1,25	2	1,625	1,625	0	0,625	1	0,625	1	0	0

Cape parrot	2	3,25	1,25	2	1,625	0	1,625	0,625	1	0	0	0,625	1
Retz's helmet-shrike	2	1,625	0,625	1	0	0	1,625	0	0	0	0	0,625	1
Buffspotted flufftail	2	4,8	1,8	3	1,6	1,6	1,6	0,6	1	0,6	1	0,6	1
African broadbill	3	3,2	1,2	2	1,066667	1,066667	1,066667	0,4	0,666667	0,4	0,666667	0,4	0,666667
Green barbet	1	9,75	3,75	6	3,25	3,25	3,25	1,25	2	1,25	2	1,25	2
African crowned eagle	1	9,6	3,6	6	3,2	3,2	3,2	1,2	2	1,2	2	1,2	2
African wood-owl	2	3,166667	1,166667	2	1,583333	1,583333	0	0,583333	1	0,583333	1	0	0
Knysna turaco	1	7	3	4	3,5	3,5	0	1,5	2	1,5	2	0	0
Livingstone's turaco	1	3,333333	1,333333	2	0	3,333333	0	0	0	1,333333	2	0	0
Black-fronted bush-shrike	2	3,25	1,25	2	1,625	1,625	0	0,625	1	0,625	1	0	0
Olive bush-shrike	2	4,75	1,75	3	1,583333	1,583333	1,583333	0,583333	1	0,583333	1	0,583333	1
Crowned hornbill	1	10	4	6	3,333333	3,333333	3,333333	1,333333	2	1,333333	2	1,333333	2
Blue-mantled crested-flycatcher	1	7	3	4	3,5	3,5	0	1,5	2	1,5	2	0	0
Tambourine dove	2	3,166667	1,166667	2	1,583333	1,583333	0	0,583333	1	0,583333	1	0	0
Orange ground-thrush	1	3,5	1,5	2	3,5	0	0	1,5	2	0	0	0	0
Spotted ground-thrush	1	10	4	6	3,333333	3,333333	3,333333	1,333333	2	1,333333	2	1,333333	2

Table A6 Risk scores of forest dependent bird species describing their susceptibility to changes in land use, calculated using the net model, using both risks and benefits to species. Reliance describes how reliant a species is on forest ecosystems, with 1 implying a low reliance and 3 a high reliance. (AT) all forests, total risk, (AF) all forests, foraging risk, (AN) all forests, nesting risk, (MT) montane forests, total risk, (LT) lowland forests, total risk, (OT) other forest types, total risk, (MF) montane forests, foraging risk, (LF) lowland forests, foraging risk, (OF) other forest types, foraging risk, (MN) montane forests, nesting risk, (LN) lowland forest, nesting risk and (ON) other forest types, nesting risk.

Taxonomic name	Reliance	AFTR	AFFR	AFBR	MFTR	LFTR	OFTR	MFFR	MFBR	LFFR	LFBR	OFFR	OFBR
Rufous-chested sparrowhawk	1	2,457143	0,857143	1,6	2,457143	0	0	0,857143	1,6	0	0	0	0
African goshawk	2	3,535714	1,285714	2,25	1,178571	1,178571	1,178571	0,428571	0,75	0,428571	0,75	0,428571	0,75
Barthroated apalis	2	1,125	0,375	0,75	0	1,125	0	0	0	0,375	0,75	0	0
Narina trogon	1	6,9	2,4	4,5	2,3	2,3	2,3	0,8	1,5	0,8	1,5	0,8	1,5
Lemon dove	2	2,4	0,8	1,6	1,2	1,2	0	0,4	0,8	0,4	0,8	0	0
Lesser spotted eagle	1	1	1	0	1	0	0	1	0	0	0	0	0
Cape batis	2	3	1	2	1,5	1,5	0	0,5	1	0,5	1	0	0
Woodward's batis	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
Barratt's warbler	3	1	0,333333	0,666667	1	0	0	0,333333	0,666667	0	0	0	0
Forest buzzard	1	1,6	0	1,6	1,6	0	0	0	1,6	0	0	0	0
Trumpeter hornbill	1	9,75	3,75	6	3,25	3,25	3,25	1,25	2	1,25	2	1,25	2
Green-backed camaroptera	3	2,5	0,833333	1,666667	0,833333	0,833333	0,833333	0,277778	0,555556	0,277778	0,555556	0,277778	0,555556
Yellowbellied greenbul	2	2,75	1	1,75	0	1,375	1,375	0	0	0,5	0,875	0,5	0,875
African emerald cuckoo	2	1,4	0,5	0,9	0	1,4	0	0	0	0,5	0,9	0	0
Southern banded snake-eagle	1	3	1	2	0	3	0	0	0	1	2	0	0
Eastern bronze-naped pigeon	2	3,333333	1,333333	2	1,666667	1,666667	0	0,666667	1	0,666667	1	0	0
Grey cuckooshrike	2	2,4	0,8	1,6	1,2	1,2	0	0,4	0,8	0,4	0,8	0	0
Chorister robin-chat	2	3	1	2	1,5	1,5	0	0,5	1	0,5	1	0	0
Forest canary	2	2,75	1	1,75	1,375	1,375	0	0,5	0,875	0,5	0,875	0	0
Olive sunbird	2	2,433333	0,833333	1,6	0	1,216667	1,216667	0	0	0,416667	0,8	0,416667	0,8
Olive woodpecker	2	5	2	3	1,666667	1,666667	1,666667	0,666667	1	0,666667	1	0,666667	1
Square-tailed drongo	2	3,25	1,25	2	0	1,625	1,625	0	0	0,625	1	0,625	1
Bearded scrub-robin	2	2,833333	0,833333	2	0	1,416667	1,416667	0	0	0,416667	1	0,416667	1
Brown scrub-robin	2	5	2	3	1,666667	1,666667	1,666667	0,666667	1	0,666667	1	0,666667	1
Grey waxbill	3	1	0,333333	0,666667	0	1	0	0	0	0,333333	0,666667	0	0
Crested guineafowl	2	2,75	1	1,75	0	1,375	1,375	0	0	0,5	0,875	0,5	0,875
Collared sunbird	2	3,6	1,2	2,4	1,2	1,2	1,2	0,4	0,8	0,4	0,8	0,4	0,8
Pinkthroated twinspot	3	1,166667	0,5	0,666667	1,166667	0	0	0,5	0,666667	0	0	0	0
Scalythroated honeyguide	3	0,811111	0,277778	0,533333	0	0,811111	0	0	0	0,277778	0,533333	0	0
Black-bellied starling	1	3,333333	1,333333	2	0	3,333333	0	0	0	1,333333	2	0	0
Bush blackcap	2	1,625	0,625	1	1,625	0	0	0,625	1	0	0	0	0
Green-backed twinspot	3	0,84127	0,285714	0,555556	0	0,84127	0	0	0	0,285714	0,555556	0	0
Mountain wagtail	1	3,25	1,25	2	0	0	3,25	0	0	0	0	1,25	2
Southern double-collared sunbird	3	0,833333	0,277778	0,555556	0	0,833333	0	0	0	0,277778	0,555556	0	0
Eurasian golden oriole	1	1,714286	1,714286	0	0,857143	0	0,857143	0,857143	0	0	0	0,857143	0
Yellow-streaked greenbul	3	1,777778	0,666667	1,111111	0,888889	0,888889	0	0,333333	0,555556	0,333333	0,555556	0	0
Yellow-throated woodland-warbler	1	8,25	3	5,25	2,75	2,75	2,75	1	1,75	1	1,75	1	1,75
Dark-backed weaver	2	3	1	2	0	1,5	1,5	0	0	0,5	1	0,5	1
Yellow-rumped tinkerbird	2	3,75	1,25	2,5	1,25	1,25	1,25	0,416667	0,833333	0,416667	0,833333	0,416667	0,833333
White-starred robin	2	2,25	0,75	1,5	1,125	1,125	0	0,375	0,75	0,375	0,75	0	0

Species													
Cape parrot	2	3	1	2	1,5	0	1,5	0,5	1	0	0	0,5	1
Retz's helmet-shrike	2	1,5	0,5	1	0	0	1,5	0	0	0	0	0,5	1
Buffspotted flufftail	2	3,6	1,2	2,4	1,2	1,2	1,2	0,4	0,8	0,4	0,8	0,4	0,8
African broadbill	3	2,4	0,8	1,6	0,8	0,8	0,8	0,266667	0,533333	0,266667	0,533333	0,266667	0,533333
Green barbet	1	9,75	3,75	6	3,25	3,25	3,25	1,25	2	1,25	2	1,25	2
African crowned eagle	1	7,2	2,4	4,8	2,4	2,4	2,4	0,8	1,6	0,8	1,6	0,8	1,6
African wood-owl	2	2,5	0,833333	1,666667	1,25	1,25	0	0,416667	0,833333	0,416667	0,833333	0	0
Knysna turaco	1	7	3	4	3,5	3,5	0	1,5	2	1,5	2	0	0
Livingstone's turaco	1	3,333333	1,333333	2	0	3,333333	0	0	0	1,333333	2	0	0
Black-fronted bush-shrike	2	3,25	1,25	2	1,625	1,625	0	0,625	1	0,625	1	0	0
Olive bush-shrike	2	3,75	1,25	2,5	1,25	1,25	1,25	0,416667	0,833333	0,416667	0,833333	0,416667	0,833333
Crowned hornbill	1	10	4	6	3,333333	3,333333	3,333333	1,333333	2	1,333333	2	1,333333	2
Blue-mantled crested-flycatcher	1	7	3	4	3,5	3,5	0	1,5	2	1,5	2	0	0
Tambourine dove	2	2,5	0,833333	1,666667	1,25	1,25	0	0,416667	0,833333	0,416667	0,833333	0	0
Orange ground-thrush	1	3,5	1,5	2	3,5	0	0	1,5	2	0	0	0	0
Spotted ground-thrush	1	10	4	6	3,333333	3,333333	3,333333	1,333333	2	1,333333	2	1,333333	2

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Table A7 Results of the GLMs disaggregated by risk. Models are abbreviated as AF (all forests), MF (montane forests), LF (lowland forests) and OF (other forest types).

Data	Model	AIC	Estimate	p	F
1. Increased abundance of small predators	Growth ~ AF	540,3253	-15,465	0,00739	7,75
	Growth ~ MF	540,6056	-15,401	0,00858	7,441
	Growth ~ LF	547,3146	-75,975	0,481	0,5043
	Growth ~ OF	547,1667	-91,896	0,424	0,6485
2. Increased fire suppression	Growth ~ AF	545,6999	-11,592	0,1532	2,099
	Growth ~ MF	542,0698	-37,84	0,0189	5,856
	Growth ~ LF	547,1547	-16,077	0,42	0,6602
	Growth ~ OF	547,8278	1,214	0,933	0,00715
3. Increased grazing pressure from domestic and wild herbivores	Growth ~ AF	543,6324	6,189	0,0451	4,209
	Growth ~ MF	543,6507	6,092	0,0456	4,19
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	547,8136	-2,821	0,886	0,02078
4. Intensified drainage management	Growth ~ AF	544,9268	3,297	0,0955	2,879
	Growth ~ MF	544,9268	3,927	0,0955	2,879
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
5. Intensified soil management	Growth ~ AF	541,3519	-39,972	0,0128	6,628
	Growth ~ MF	541,3519	-39,972	0,0128	6,628
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
6. Intensified thinning	Growth ~ AF	545,092	3,499	0,105	2,711
	Growth ~ MF	545,092	3,499	0,105	2,711
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
7. Reduced abundance of canopy species	Growth ~ AF	544,1535	5,275	0,0607	3,669
	Growth ~ MF	544,1535	5,275	0,0607	3,669
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
8. Reduced rotation length (including fragmentation effects)	Growth ~ AF	544,0699	9,398	0,0579	3,756
	Growth ~ MF	544,0699	9,398	0,0579	3,756
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
9. Removal of deadwood	Growth ~ AF	542,8914	-25,364	0,0298	4,984
	Growth ~ MF	542,8914	-25,364	0,0298	4,984
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
10. Reduced area of forest	Growth ~ AF	546,3402	-7,554	0,232	1,461
	Growth ~ MF	546,3402	-7,554	0,232	1,461
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
11. Reduced diversity of tree species	Growth ~ AF	542,0698	-37,84	0,0189	5,856
	Growth ~ MF	542,0698	-37,84	0,0189	5,856
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
12. Increased forest fires	Growth ~ AF	546,3402	-7,554	0,232	1,461
	Growth ~ MF	546,3402	-7,554	0,232	1,461
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
13. Loss of habitat through urbanisation	Growth ~ AF	546,3402	-7,554	0,232	1,461
	Growth ~ MF	546,3402	-7,554	0,232	1,461
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
14. Increased selective logging	Growth ~ AF	545,7101	10,467	0,154	2,089
	Growth ~ MF	545,7101	10,467	0,154	2,089
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
15. Increased plantations	Growth ~ AF	544,9198	6,278	0,0951	2,886
	Growth ~ MF	544,2187	6,803	0,063	3,602
	Growth ~ LF	546,6658	-24,85	0,291	1,14
	Growth ~ OF	547,701	-8,481	0,72	0,1295
16. Increased invasion by alien trees	Growth ~ AF	547,4048	2,369	0,521	0,4166

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	Growth ~ MF	547,315	3,587	0,481	0,5039
	Growth ~ LF	547,6972	4,642	0,717	0,1332
	Growth ~ OF	547,8273	1,131	0,931	0,00758
17. Increased fragmentation	Growth ~ AF	547,1618	2,045	0,422	0,6533
	Growth ~ MF	547,239	1,975	0,45	0,5779
	Growth ~ LF	546,7533	25,4085	0,309	1,053
	Growth ~ OF	547,6996	-8,957	0,719	0,1309
18. Local harvesting (muthi harvesting)	Growth ~ AF	547,2073	-5,552	0,439	0,6089
	Growth ~ MF	547,2073	-5,552	0,439	0,6089
	Growth ~ LF	545,8352	NA	NA	NA
	Growth ~ OF	545,8352	NA	NA	NA
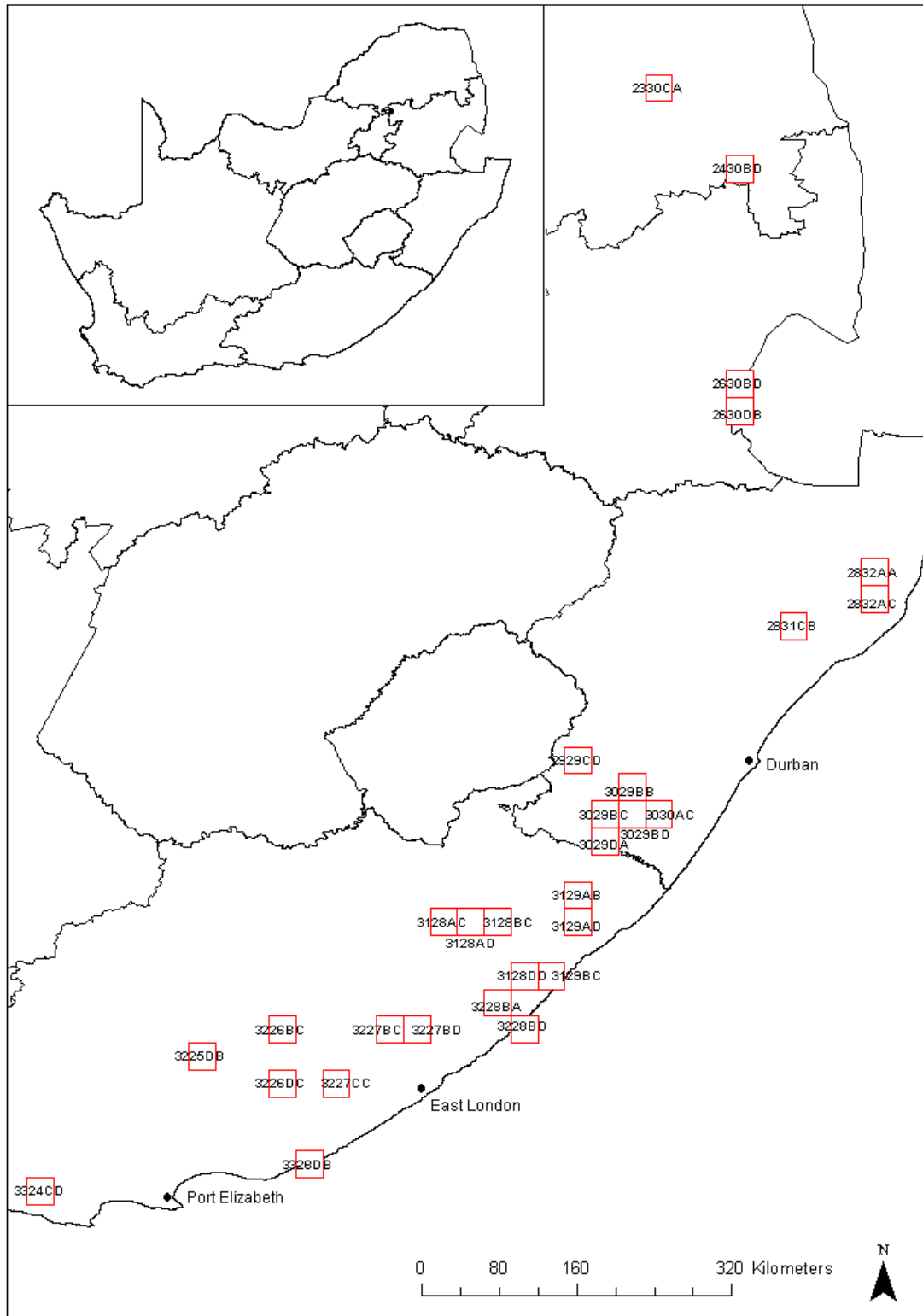
Figure A1 The thirty sites, with more than ten forest dependent bird species lost over the twenty year period between the first and second South African Bird Atlas Project, analysed in this study to determine patterns between changes in species occupancy and land cover over the same period.