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An evaluation of bird presence and breeding activity in regenerating coastal dune forests, Maputaland, South Africa

by

Martin Russell Taylor

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ABSTRACT

Researchers use changes in bird community composition to illustrate ecological succession in restoration studies. These studies utilise a variety of methods, some of which may bias their outcomes. Here I used the point and line transect methods to obtain community compositions as well as to estimate total density, species richness and species diversity for bird communities living in a successional sere of coastal dune habitats. The two methods yielded similar estimates of community variables in the younger regenerating sites with the point transect method giving higher estimates in the older regenerating sites. The line transect method provided higher density estimates than the point transect method across all sites but there was no significant difference between variance estimates for the two methods. The point transect method reached the maximum number of detectable species in half the time that it took the line transect method. In contrast to my expectations, the point transect method proved to be superior in terms of efficiency.

Past research suggests that bird presence itself does not reflect on colonisation success. I compared the community composition, variables and age related trends of point transect and breeding assemblages within the study area. The breeding assemblages represented a subset of the point transect assemblage. Age related trends were significantly similar with both assemblages approaching the variables noted on an undisturbed reference site with an increase in regeneration site age. I concluded that presence based surveys of bird communities adequately captured age related trends to make valid conclusions about patterns of succession.

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A number of people need to be thanked for the assistance that they provided during the process of completing this study. Most importantly I am grateful to Professor Rudi van Aarde who gave me an opportunity to undertake this study as well as providing invaluable insights into succession and ecology in general. Thanks must also go to my co-supervisor, Dr Theo Wassenaar, who taught me the importance of sitting in a coffee shop with a pen and paper as well as the often quoted mantra borrowed from Professor Stuart Pimm….what is the problem, why it is important and what are we doing to solve it.

This study would have not been possible without the support and cooperation of Richards Bay Minerals and especially Rynhard Kok, superintendent of the rehabilitation programme at Richards Bay Minerals, who provided advice and guidance in the field. My research was funded by the National Research Foundation, Richards Bay Minerals and the University of Pretoria through core grants to Professor van Aarde. The study was conducted under the auspices of the Conservation Ecology Research Unit (CERU), University of Pretoria.

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CHAPTER 1

INTRODUCTION

GENERAL OVERVIEW

While the world's economies have enjoyed unprecedented expansion, ecological systems have suffered rapid degradation (Perrow and Davy 2002). Human activities have negatively affected virtually all ecosystems (Wilson 1999). From an ecological perspective, restoration, as a conservation tool, is one of the only sensible land use options left for damaged natural ecosystems (Pimm 2001). The need for the restoration of damaged ecosystems has given rise to the discipline of restoration ecology (Davis and Slobodkin 2004). Ecological restoration is the process of assisting the recovery and management of ecological integrity including the recreation of both structure and functions of an ecosystem. The goal of a restoration ecologist is therefore to understand and assist in the recovery of an ecosystem that has been degraded, damaged or destroyed (Davis and Slobodkin 2004). Restoration of ecosystems or landscapes sometimes relies on natural succession (Luken 1990) and many case studies support this perspective (e.g. van Aarde *et al.* 1996, MacMahon 1997). Understanding succession (in the context of restoration) is therefore of great importance to restoration ecologists.

Perrow and Davy (2002) state that bird community parameters, such as species diversity and composition accurately reflect the major habitat changes due to succession and potentially, on restoration success or failure. While this may be the case, researchers can only make inferences if the methods of data collection return reliable information. After all, flawed procedures produce incorrect observations and any conclusions drawn from these observations will be incorrect (Wiens 1992). In addition to being robust, the most suitable method must be efficient in terms of the amount of time required by the observer to adequately sample study sites. Given the importance of using the most suitable methods during surveying events, my thesis focuses on 1) comparing two routinely used distance methods to determine which is the most efficient for my study area and 2) determine if patterns of succession modelled from the observed bird assemblages agrees with that derived from breeding assemblages. By doing this I intend to contribute to the discipline of restoration ecology and in particular the role that bird communities play in studies of succession.

THE COASTAL DUNE RESTORATION PROGRAMME OF RICHARDS BAY MINERALS

The opencast mining of heavy minerals in the coastal dunes north of Richards Bay started in 1977 (Camp 1990). The initiation of this rehabilitation programme coincided with the inception of mining and since then has resulted in the development of a sere of known-age regenerating sites in the trail of the mining paths. The mining company starts the rehabilitation process by spreading topsoil and controlling wind erosion after having reshaped the dunes to mimic shapes and contours typical of the regions unmined dunes. From this point, regeneration occurs through a process of natural succession that is driven by unaided colonisation and replacement (van Aarde *et al.* 1996). The sites that I studied during 2004 lie along a northeast to southwest axis and range from one year old grasslands to 28 year old woodlands dominated by the pioneer tree species Sweet thorn (*Acacia kosiensis*) (formerly *Acacia karroo*). In addition, the relatively undisturbed coastal dune forest to the north and east of the regenerating sites formed part of my study area. This area of regenerating coastal dune forest has given ecologists a unique opportunity to study various ecological concepts associated with succession (e.g. van Aarde *et al.* 1996, Davis *et al.* 2003, Redi *et al.* 2005, Wassenaar *et al.* 2005).

RAPID ASSESSMENT OF BIRD COMMUNITIES

Researchers utilise a wide suite of rapid assessment methods to collect information from bird communities (e.g. Bibby *et al.* 1993, Buckland *et al.* 2001, and O’Dea *et al.* 2004). Earlier surveys in my study area used the line transect method (Kritzinger and van Aarde 1998, Niemand 2001, Wassenaar *et al.* 2005). However, studies elsewhere (Bibby *et al.* 1993, Blake and Loiselle 2001, Shankar Raman and Mudappa 2003, Naidoo 2004) preferred the point transect method for surveys in dense habitat, such as those encountered along the coastal dunes of the study area. Both methods have advantages and disadvantages (Buckland *et al.* 2001). In addition, these two methods, like other methods, might give different estimates of the variables within a community (Blake and Loiselle 2001).

The estimates obtained from data obtained from a survey method must be robust as well as the method being efficient. It is thus important to know if the two most widely used distance methods differed in their ability to detect bird species. Furthermore, it is important to understand how data collected using these methods influences abundance estimates and temporal trends in community variables. More specifically, in the first part of the thesis I ask if 1) data collected using the two methods give different estimates of community variables and 2) which of the two methods is more time efficient in estimating community variables?

MEASURING THE SUCCESS OF RESTORATION PROGRAMMES

Restoration ecologists have used bird communities as surrogates of habitat recovery in studies of succession worldwide (e.g. Dobkin *et al.* 1998, Canterbury *et al.* 2000, Freshman Browder *et al.* 2002). Two such studies within my study area have focussed on age related trends and succession (Kritzinger and van Aarde 1998, Wassenaar *et al.* 2005). Kritzinger and van Aarde (1998) found that species richness and diversity changed bimodally across the sere of rehabilitating sites as well as an almost complete turnover in species composition. Wassenaar *et al.* (2005) evaluated restoration success by measuring the rate at which the community composition of various taxa converged to those reflected at an undisturbed reference site. For birds, they showed that similarity indices increase with time since disturbance.

These studies, like most rapid assessments of bird communities, assigned presence (and abundance) to a species based on the observation of one or more individuals. However, the presence of a species in an area does not necessarily equate to the species having successfully colonised that area. Such individuals could be resident (i.e. breeding), in transit, or might be using the site only for foraging or shelter (Owiunji and Plumptre 1998, Blake and Loiselle 2001, Dranzoa 2001). Their presence therefore may be misleading and thus raises the question as to whether conclusions based upon bird presence data are sufficient to infer successional development in response to a rehabilitation programme. This has implications for the perceived success of rehabilitation efforts. In the second part of my study, I therefore ask if 1) the community compositions and community variables of the observed assemblage and breeding assemblage are similar, 2) the composition of the nesting assemblage changes over time in a similar manner to that of the observed assemblage, 3) species that occurred at higher density are more likely to be nesting and 4) if the presence of a bird species during distance sampling signifies colonisation deduced from nesting surveys. I answer these questions in Chapter 4 by differentiating between the point transect assemblage and the resident breeding assemblage. In the final Chapter, I synthesise the results of my research.

CHAPTER 2

THE STUDY AREA

The study took place at the southern end of the Maputaland-Pondoland regional centre of plant diversity (van Wyk 1996). I conducted my surveys along a narrow belt of coastal dune forests that stretches northwards from 25km northeast of the coastal town of Richards Bay (28°43'S, 32°12'E) to the Sokhulu Forest (28°40'S, 32°14'E). The width of the study area varied from 1.5 to 2.5 km (Figure 1).

These coastal dune forests cover a belt of fixed Quaternary heavy mineral bearing sand dunes that run parallel to the Indian Ocean. The dunes rest on a Pleistocene beach underlain by the Port Durnford Formation. The latter rests on the sediments of the Zululand Group (Wilson and Anhaeusser 1998). The substrate of the dunes is <20 000 years old, with the forests themselves being considerably younger (<10 000 years old) (see references in Lawes 1990).

The climate of the region is subtropical, hot and humid, with high summer temperatures - mean values exceed 30°C for about 26 days per year (Kritzinger 1996). Winters are moderate with the lowest mean monthly temperature being 13.4°C. Sixty percent of the total precipitation occurs between October and March over the summer months (Niemand 2001). Relative humidity is high in summer, with a monthly mean of 84% at 08h00 and 68% at 14h00. Humidity drops to below 70% in winter (Kritzinger 1996). Based on rainfall and temperature profiles, Ferreira (1997) identified January and February as representing summer (hot, wet period), while June, July and August represent winter (cool, dry period). The months between these periods represent autumn (March, April and May) and spring (September, October, November, December) (both, hot dry periods) (Ferreira 1997). The information in Table 1 reflects average maximum and minimum temperatures recorded at the Richards Bay Minerals Ecology Centre as well as average monthly rainfall recorded at the Richards Bay Weather Station, both of which are located close to my study area.

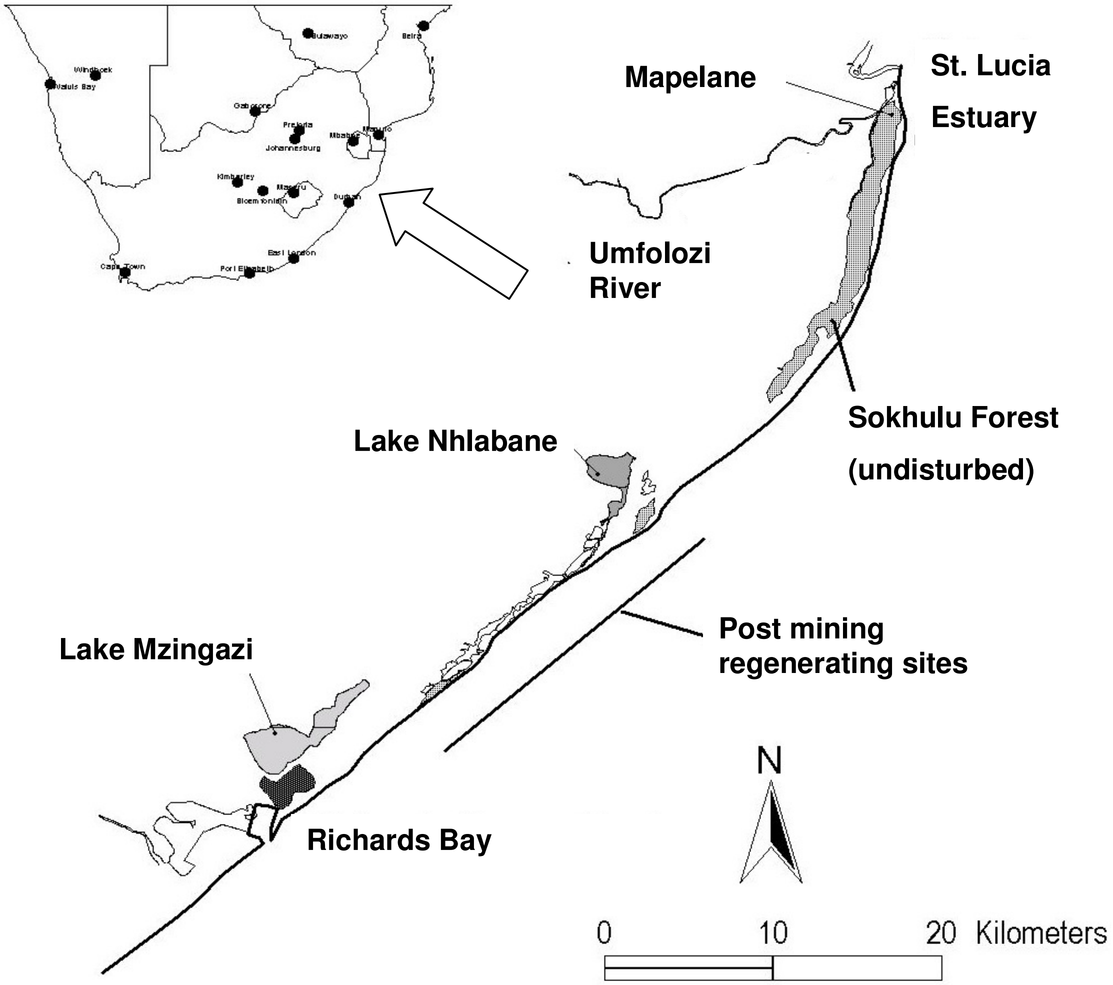

Figure 1. A line diagram of the study area in relation to its location in South Africa. The diagram also shows the location of the regenerating sites and the undisturbed reference site (Sokhulu Forest) in relation to the major rivers, lakes and the city of Richards Bay.

Table 1. Mean ±SD (standard deviation of the mean) maximum and minimum monthly temperatures recorded at the Richards Bay Weather Station (28°43'S, 32°12'E) from 1970 to 2004 and mean ±SD monthly rainfall recorded at the Ecology Centre of Richards Bay Minerals from 1976 to 2004 (28°39'S, 32°19'E).

Month	Maximum temperature (°C)	Minimum temperature (°C)	Monthly rainfall (mm)
January	29±1	21±1	153±96
February	29±1	21±1	205±176
March	28±1	20±1	152±105
April	26±1	18±1	136±92
May	25±1	15±1	103±78
June	23±1	12±1	89±67
July	22±1	12±1	105±65
August	24±1	14±1	80±72
September	25±1	16±1	126±171
October	25±1	17±1	125±56
November	27±1	19±1	163±121
December	28±1	20±1	109±76

Historically, subsistence agriculture accounted for large-scale clearance of the northern part of the study area (Weisser and Marques 1979). More recently, open cast dune mining (van Aarde *et al.* 1996) has been the cause of disturbance of relatively large portions of dune forest.

In 1968, the Industrial Development Corporation (IDC) undertook detailed exploration in the study area. In 1976, the IDC and QIT (Quebec Iron and Titanium) formed Richards Bay Minerals (RBM) in order to mine the mineral rich sands of the coastal dunes (Wilson and Anhaeusser 1998) with mining operations starting in July 1977 (Camp 1990). The first step of the mining process involves the removal of vegetation in front of the mine path followed by the collection of topsoil that is stockpiled for later use (van Aarde *et al.* 1996). Dredgers floating on a 50ha artificial pond use heavy-duty water jets to slump the dune wall in front of the pond. Dredged material passes through a floating concentrator that separates the heavy mineral (5% of the volume) from the bulk and discards the rest (Camp 1990).

The rehabilitation programme started soon after mining (Camp 1990). Recontoured sand approximates the topography of dunes prior to mining operations. Previously collected topsoil is spread over these dunes and is sown with a mixture of exotic and indigenous annuals. Protective ~1.5m high windbreaks are erected at right angles to prevailing winds to minimize the impact of wind erosion. The grassland that develops on these dunes constitutes the initial vegetation type. Following this initial assistance, rehabilitation management is limited to fire protection and the control of exotic vegetation (van Aarde *et al.* 1996).

The topsoil used here is rich in Sweet thorn seeds (Wassenaar and Taylor 2004). As a pioneer species, Sweet thorn seedlings establish soon after the spreading of the topsoil (van Aarde *et al.* 1996). Broad-leaved pioneer tree species, such as White milkwood (*Sideroxylon inerme*), White ironwood (*Vepris lanceolata*) and Coast silver oak (*Brachylaena discolor*) start colonising from about 12 years after the initiation of rehabilitation activities (van Aarde *et al.* 1996). In general, the number of tree species increases and the number of pioneer species decrease with regeneration age across the sere of rehabilitating sites (van Aarde *et al.* 1996). As site age increases, the densities of mid-successional dune forest species such as Coastal red milkwood (*Mimusops caffra*), White stinkwood (*Celtis africana*) and Pigeonwood (*Trema orientalis*) increases. Consequently, the rehabilitation programme has given rise to a sere of regenerating sites orientated along a north to south axis. At the onset of my study in 2004 the oldest site, located in the southern portion of the lease area, was ~28 years old while the youngest site (one year old in 2004) was located to the north of the Nhlabane estuary (Figure 1). By then the regenerating sites covered 731 hectares (Rynhard Kok pers. comm.[1]).

In the study area, the rehabilitation of indigenous dune forests occurs on one third of the mining lease area. As a condition of the mining license, RBM also establishes Beefwood (*Casuarina equisetifolia*) plantations on the remaining two-thirds of the post-mined areas. This fast growing commercial species provides fuel for the local charcoal industry and stands thereof covered 1235 hectares in 2004. Inland of the Beefwood plantations is a mosaic of rural residential areas interspersed with subsistence farming plots and Bluegum (*Eucalyptus grandis*) plantations (Appendix 1). Exotic weeds such as Lantana (*Lantana camara*) and Guava (*Psidium guajava*) occur in extensive stands in both vegetation types. The eastern coastal side of the regenerating sites is bordered by a 200m to 400m wide strip of unmined vegetation, colloquially known as the ecostrip.

The outermost ~100m of the ecostrip lies on the coastal foredune and is characterised by dense, stunted vegetation. Typical species include Veld fig (*Ficus burtt-davyi*), Coastal red milkwood, Coastal silver oak and White pear (*Apodytes dimidiata*) (Weisser 1980). The mining company leaves the ecostrip intact as a condition of the mining license with the intention that it would act as a source area for plant and animal populations (Redi *et al.* 2005).

The Sokhulu forest is relatively undisturbed and served as the reference site for this study (Redi *et al.* 2005). It is located about 20km northeast of the regenerating sites and is contiguous with the Mapelane Forest Reserve (Figure 1). Woody species such as Coastal red milkwood, White

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[1] Rynhard Kok; Superintendent: Rehabilitation, P.O. Box 401, Richards Bay Minerals, 3900, South Africa.

stinkwood, Small leaved jackalberry (*Diospyros natalensis*) and Buffalo thorn (*Ziziphus mucronata*) dominate the upper canopy. Horsewood (*Clausena anisata*), Buckweed (*Isoglossa woodii*) and Large forest asparagus (*Asparagus falcatus*) dominate the lower canopy. The Mapelane/Sokhulu complex is located immediately to the south of the Umfolozi/Lake St Lucia estuary (Wassenaar 2004). I provide further detail of the vegetation of the study area in Appendix 1.

De Klerk *et al.* (2002) analyzed the biogeography of birds endemic to the Afrotropical mainland. Their Tongaland–Pondoland Biogeographical Province is part of a Southern Savannah Subregion and includes the Indian Ocean coastal belt and all miombo woodlands. It also coincides roughly with the Maputaland Centre of Endemism as delineated by van Wyk (1996). Three hundred and ninety three species occur in the Tongaland-Pondoland Province and a number of these are endemic to sand forest (de Klerk *et al.* 2002). The 138 species that I recorded in my study area are listed in Appendix 2.

CHAPTER 3

THE USE OF DISTANCE METHODS TO EVALUATE THE BIRD COMMUNITIES OF REGENERATING COASTAL DUNE FOREST IN MAPUTALAND, SOUTH AFRICA

INTRODUCTION

Researchers can estimate the species richness, diversity, densities, and or relative densities of birds present during surveys from data collected utilising a variety of methods, including the family of distance methods (Bibby *et al.* 1993). Different sampling methods, however, may yield different data and subsequently influence estimates of population and community variables (Blake and Loiselle 2001). The choice of distance sampling method may thus unintentionally influence inferences made on, for instance, the post-mining recovery of bird communities in the regenerating dune forests. According to Buckland *et al.* (2001), distance methods may differ in the number of individual birds that can be detected, the ease with which cryptic birds can be noted in dense vegetation, and the densities that are estimated from data collected during surveys. For the rapid assessments of bird communities, the preferred method must provide data from which robust estimates can be derived as well as being efficient in its ability to saturate sampling efforts within a study area.

Point transects (Bibby *et al.* 1993, Blake and Loiselle 2001, Buckland *et al.* 2001, Shankar Raman and Mudappa 2003, Naidoo 2004) and the line transect (Bibby *et al.* 1993, Buckland *et al.* 2001), two sampling techniques belonging to the distance family, have both been widely used for bird surveys. Detectability decreases with an increase in distance away from an observer (Buckland *et al.* 2001). To solve this problem detection curves are fitted to estimate the effective area sampled and subsequently the densities of species within a site-specific community. The distance methods are based on several assumptions (see Buckland *et al*. 2001). These include the assumption that one detects all birds within the sampling area and at their initial locations. Movement in response to an observer moving down a line transect may confound this assumption (Bibby *et al*. 1993), as would disturbance generated by an observer approaching a point transect station (Buckland *et al*. 2001). Further assumptions are that distance measurements from the observer to the bird must be accurate and that individual birds are detected independently of each other.

A line transect survey takes place along pre-arranged straight lines that do not cross each other and that are far enough apart to ensure that birds are detected independently of each other. Line transects are most suitable for habitats that are relatively uniform. Line transects may also be more accurate than point transects (Bibby *et al*. 1993) due to the area sampled with point counts increasing geometrically with distance from the observer, but only linearly with transects (Verner and Ritter 1983). The line transect method also has disadvantages. In dense habitats, (typical of the study site), observers may struggle to cross the terrain. Birds may be disturbed whilst walking which would confound the assumption that birds are detected at their initial location.

Point transect surveys differ from line transect surveys in that the observer is stationary and records individuals in a radius around him. Songbird surveys often utilise point transects rather than line transects for several reasons (Bibby *et al*. 1993). Firstly, the observer can concentrate on detecting, locating, and identifying birds without the need to traverse difficult terrain (Bibby *et al.* 1993, Buckland *et al*. 2001). Secondly, the observer can take the easiest route to and from a point (Buckland *et al*. 2001). Finally, direct distances are easier to estimate than the perpendicular distances used in line transect sampling (Buckland *et al.* 2001). Point transects do however, have disadvantages. For instance, an observer may disturb birds when approaching a point (Buckland *et al*. 2001). Point transect sampling is also more sensitive to bird movement than is line transect sampling (Buckland *et al.* 2001). In addition, the point transect method produces much larger errors with violations of the assumptions central to distance sampling than the line transect method (Verner and Ritter 1983).

Both methods therefore have advantages and disadvantages. In this study I compare the two methods in their ability to provide data based upon bird communities residing in nine different habitats. In order to do this, I compared density, community composition and community structure (Shannon diversity and Margalef's species richness) derived using two different sampling procedures applied across eight sequentially aged regenerating coastal dune forest sites and an undisturbed reference site. More specifically I asked the following questions:

1) Do the two survey methods give different estimates of community composition and community variables (species richness and diversity)?
2) Which of the two methods is more efficient in collecting data used to estimate community variables?

Based on Blake and Loiselle's (2001) observation that different sampling methods influence estimates of community variables, I expect that the two distance methods, namely the point and line transect methods, will give different estimates of community variables. Furthermore, I expect that the point transect method will be more efficient than the line transect in its ability to saturate sampling effort within a site due to its perceived ability to detect more species (Bibby *et al.* 1993, Buckland *et al.* 2001).

MATERIALS AND METHODS

Data collection

I sampled eight known age regenerating sites and a reference site (see Appendix 1). All bird species except raptors, water birds (e.g. ducks, geese and storks) and aerial foragers (swifts and swallows) were included in the study (Kritzinger 1996, Naidoo 2004). I treated the Yellow weaver (*Ploceus subaureus*) and the Lesser-masked weaver (*P. intermedius*) as one morpho-species due to the difficulty of distinguishing between the females of these species whilst in a flock (Sinclair and

Hockey 1996).

Point and line transect data were collected from 3 January to 15 March 2005. Surveying took place from 20 minutes before sunrise to 4 hours thereafter on days when it did not rain or when it was not exceptionally windy. The data collected included the locality of the line transect/point transect, time spent surveying, species noted, sighting angle and distance to the bird. I measured distances to where the bird was first seen using a range finder. Initially, I used a "Ranging" range finder (calibrated at 0.5-meter intervals from 9m to 13m, 1m intervals from 13m to 40m, 2m intervals to 60m and 5m intervals to 75m). Later on I used an Impulse 200LR laser range finder (Laser Technology Incorporated, 7070 South Tucson Way, Centennial, Colorado, 80112, USA) calibrated at 0.01m. I measured distance to a point vertically below the bird and not to the bird itself. I placed birds detected solely by sound (aural detection) into 10m intervals. In addition, I measured bearings to birds to the nearest 5° with a Suunto prismatic compass (Suunto Finland, Valimotie 7, FIN -01510, Vaanta, Finland). The sequence of sampling across sites and between methods was randomised and species accumulation curves were constructed to determine adequate sampling effort. Adequate sampling effort was defined as the point where the rate of increase of the species accumulation curve per site was <0.1. The amount of time that I spent surveying served as an indication of sampling effort. I did not record the time that it took to get to sampling sites or the time it took to lay out the line transects.

Line transects were established in an east to west direction and perpendicular to the slope of the dunes. All transects were 300m in length and at least 200m apart. I included detections made within 60m of either side of the transect. I determined the number of line transects within a site based on the area and shape of the site. I surveyed all transects within a site at least three times with a minimum of four days between sampling sessions.

The locations of point transect stations within each site were at least 150m apart to ensure independence of observations (Buckland *et al.* 2001). Sampling at each of these points occurred over an 8-minute period. To reduce disturbance, I waited 2-minutes after arriving at the site before sampling. I recorded all birds that I saw or that I could hear within a 60m radius.

Analysis

I standardised sampling effort by pooling randomised point transect sampling units into groups so that the time required to complete a point transect group was similar to the average time taken to complete a line transect. This enabled direct comparisons of information extracted from the two sampling protocols.

For each survey method I calculated the total number of species, Shannon diversity (H'), and Margalef's species richness index for each of the sampling sites using Primer V5 (Clarke and Gorley 2001). I used DISTANCE 4.1 (Buckland *et al.* 2001) to calculate the relative density of birds within a site for both methods. DISTANCE 4.1 requires a minimum of 60 individuals per species per sample. In order to satisfy this requirement I combined all individuals of different species as one group for the purposes of the analysis. I applied several models of the probability density function (half normal, uniform, hazard with cosine or polynomial adjustment terms) to the data and selected the best-fit detection models using Akaike Information Criteria. A density of individual birds per hectare was obtained from which I could calculate the relative density of individual species by using the species specific contribution to the total group. Each set of community variables were plotted as a function of the regeneration age of the sites where regeneration age was the number of years that had elapsed since rehabilitation of the relevant sites commenced. I utilised an unpaired t test to compare the means of the community variables obtained from the two methods.

I spatially represented interrelationships of community compositions within sites and between sites using non-metric multi dimensional scaling (NMDS) (see McCune and Grace 2002). Relative abundance data were log transformed to reduce the bias introduced by a few common species to similarity. A cluster analysis distinguished between assemblages detected by the point and line transect methods (Primer V5) at a 75% similarity. I used a two way nested ANOSIM to test whether there was significant differences between method types within habitat type.

I used Estimate S (http://viceroy.eeb.uconn.edu/EstimateS) to generate sample-based rarefaction curves (species accumulation curves) using the analytical formulas of Colwell *et al.* (2004) and Mao *et al.* (2005) for both methods within a site. I defined saturation as the point where the rate of species accumulation over five sample intervals fell below 0.10. The point at which saturation occurred for both methods was taken as a measure of efficiency. I calculated the variance for community variables for both the point and line transect methods within all study sites and compared variances using an Fmax test (Rosner 2005). Here P-values <0.05 suggest significant differences.

RESULTS

The 139 line transects, completed across eight regenerating sites and an undisturbed site, yielded 1888 observations and 58 species during 57.1 hours of survey. The 223 point counts, conducted across these nine sites yielded 62 species among 1729 observations during the 30.4 hours that were required to saturate sampling effort (Table 2). I took saturation as the point at which the rate of species accumulation over five samples within a site fell below 0.10. On all sites saturation of species accumulation curves occurred faster for the point transect method than the line transect method (Figure 2).

Table 2. The number of bird species recorded during two different distance sampling procedures and the time that it took to saturate species accumulation curves. The area and age of each site is also presented. The age of the undisturbed reference site is not known.

Site age (years)	Area (ha)	Number of species		Time taken to saturate sampling effort (minutes)	
		Point transect	Line transect	Point transect	Line transect
1	39	17	19	168	360
4	32	19	19	168	312
8	104	16	19	144	288
12	87	16	17	144	288
16	124	15	11	192	312
20	80	18	14	192	264
24	190	24	20	192	408
28	76	34	29	256	512
Unknown	500	39	38	288	576

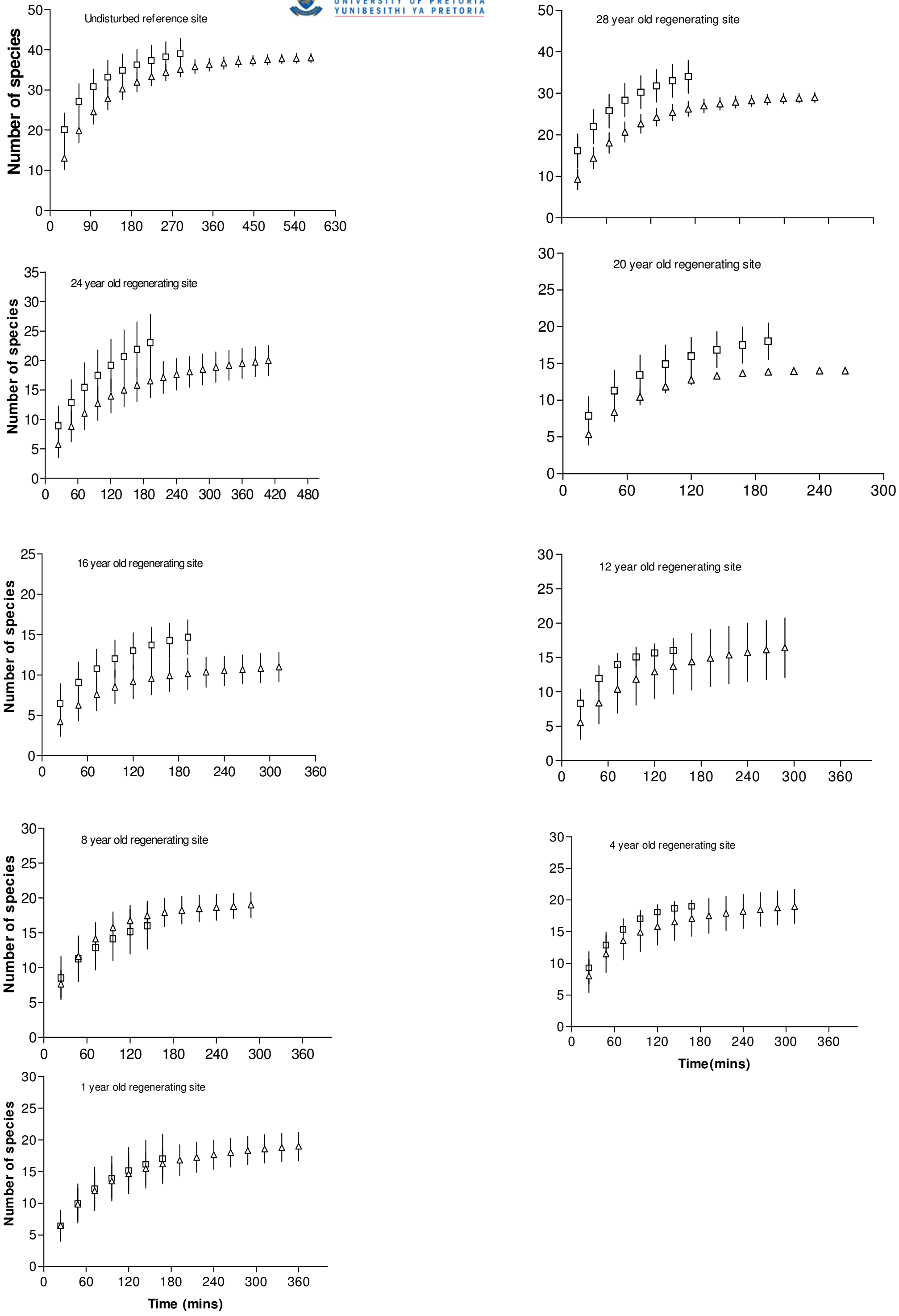

Figure 2. Species accumulation curves for each of the study sites. I defined saturation as the point at which the rate of increase was less than 0.1 per sampling unit. Accumulation curves based on data from point transects saturated sooner than those from line transects for all sites. Squares indicate point transect groups and triangles represent the line transects. Points are mean values calculated by the Moa Tau method (http://viceroy.eeb.uconn.edu/EstimateS).

Community variables

The cluster analysis revealed six groupings of assemblages (Figure 3). Assemblages defined by the two methods occurred within the same groupings. The two way nested ANOSIM indicated that there was no significant difference between the two methods within a site (R = 0.136, *P* = 0.179) but that there was a significant difference between the assemblages across the nine sites (R = 0.185, *P* = 0.001).

The Shannon diversity indices (H') for the point and line transects showed a bimodial distribution along the chronosequential sere (Figure 4). Values for both methods peaked at the unmined reference site and the 8 year old site. More importantly, Shannon diversity estimates were similar for the 1 to 8 year old regenerating sites. The point transect method gave significantly higher estimates for the 20 to 28 year old regenerating sites and the reference site (Figure 4). There was no significant difference in the amount of variance for Shannon estimates obtained from the two methods with the exception of one site (Table 3). Estimates of Margalef's species richness based on the two different sampling procedures followed similar trends (Table 4) to the Shannon estimates. There was no significant difference in the amount variance around the mean for species richness estimates of both methods with the exception of one site (Table 4). The line transect method yielded higher densities than the point count method for all sites (Figure 2). The amount of variance surrounding the mean density values for the two methods did not differ significantly (Table 5).

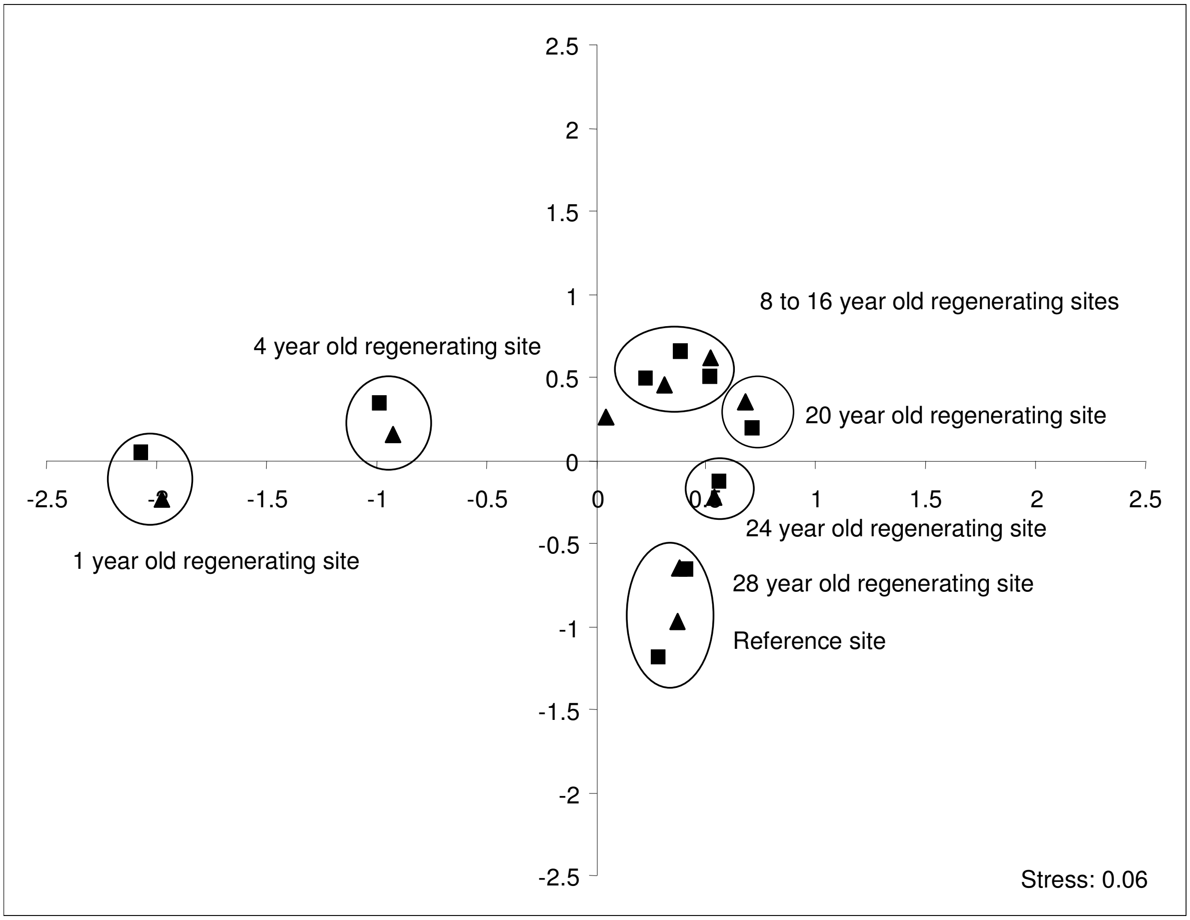

Figure 3. A non-metric multidimensional scaling (NMDS) ordination of community compositions derived from line transects (triangle) and point transects (squares) per site. I used Bray Curtis distances calculated from fourth-root transformed relative abundances. Stress value was 0.06. A solid black line encircles the locations of each assemblage in multi dimensional space. Communities were grouped using a cluster analysis at 75% similarity.

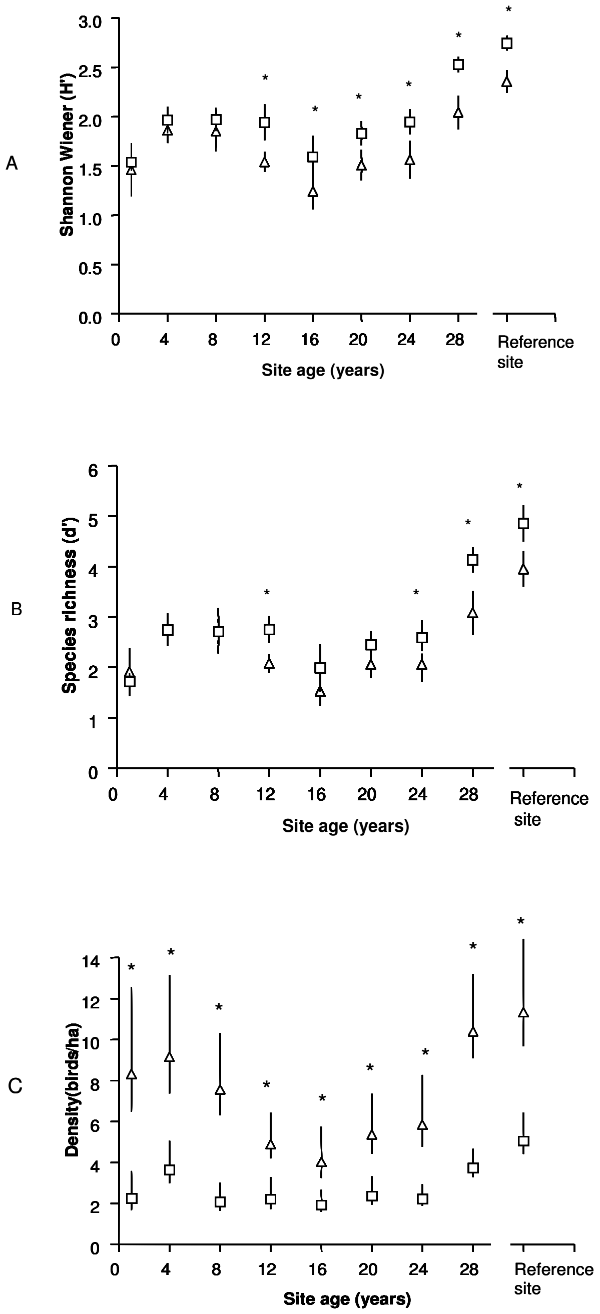

Figure 4. The Shannon diversity (H') (A), Margalef's species richness (B) and density estimates of birds per hectare (C) derived from data collected during line transects (triangles) and point transects (squares) on eight different-aged rehabilitating sites and the undisturbed reference site. Vertical lines represent 95% confidence limit estimates. Asterisks indicate a significant difference at P<0.05 between mean values. Sample sizes are presented in Table 3 and 4.

Table 3. Unpaired t-test and *P* values comparing the mean Shannon diversity (H') for the point and line transect methods. I used an Fmax test to test if the variances are significantly different. Asterisks represent sites where the test statistic suggests a significant difference at P<0.05.

Site age (years)	Shannon Diversity Mean ± SE (n)		t value$_{df}$	*P* value (t test)	Fmax $_{df1\ df2}$	*P* value (Fmax)
	Point transect	Line transect				
1	1.54±0.149 (7)	1.46±0.432 (15)	0.472_{20}	0.642	$0.009_{14,6}$	0.077
4	1.97±0.223 (7)	1.86±0.212 (13)	1.031_{18}	0.316	$0.434_{12,6}$	0.409
8	1.97±0.193 (6)	1.85±0.328 (12)	0.815_{16}	0.427	$0.098_{11,5}$	0.125
12	1.94±0.246 (6)	1.54±0.167 (14)	4.292_{18}	0.0004*	$0.131_{13,6}$	0.119
16	1.59±0.346 (9)	1.24±0.300 (13)	2.521_{20}	0.0203*	$0.050_{12,8}$	0.317
20	1.83±0.198 (8)	1.51±0.252 (11)	2.602_{16}	0.019*	$0.419_{10,7}$	0.249
24	1.95±0.206 (9)	1.56±0.311 (17)	2.962_{23}	0.007*	$0.460_{16,8}$	0.129
28	2.53±0.130 (8)	2.04±0.277 (16)	4.679_{22}	<0.01*	$0.075_{15,7}$	0.025*
Unknown	2.73±0.123(9)	2.36±0.187(18)	5.671_{25}	<0.01*	$0.453_{17,8}$	0.116

Table 4. Mean (±SE) Margalef's species richness as a function of site ages and based on information collected during point and line transect methods. I report the results of the comparison of means and use an Fmax test to test whether the variances are significantly different. The degrees of freedom are presented as footnotes and values in brackets denote sample sizes. Asterisks represent significant differences at P<0.05.

Site age (years)	Margalef's species richness Mean ± SE (n)		t value($_{\text{df}}$)	*P* value (t test)	Fmax($_{\text{df1 df2}}$)	*P* value (Fmax)
	Point transect	Line transect				
1	1.72±0.274 (7)	1.91±0.764 (15)	0.633 $_{20}$	0.534	7.749$_{14,6}$	0.009*
4	2.74±0.499 (7)	2.76±0.483 (13)	0.096 $_{18}$	0.924	1.065$_{6,12}$	0.434
8	2.71±0.396 (6)	2.73±0.721 (12)	0.054 $_{16}$	0.958	3.322$_{11,5}$	0.098
12	2.76±0.428 (6)	2.08±0.295 (14)	4.078 $_{18}$	<0.01 *	2.096$_{5,13}$	0.131
16	1.99±0.756 (9)	1.53±0.449 (13)	1.809 $_{20}$	0.086	2.843$_{8,12}$	0.050
20	2.45±0.440 (8)	2.06±0.415 (11)	1.987 $_{17}$	0.063	1.124$_{7,10}$	0.419
24	2.59±0.552 (9)	2.06±0.546 (17)	2.345 $_{24}$	0.028 *	1.020$_{8,16}$	0.460
28	4.14±0.408 (8)	3.08±0.703 (16)	3.888 $_{22}$	<0.01*	2.974$_{15,7}$	0.075
Unknown	4.86±0.576 (9)	3.96±0.578 (18)	3.885 $_{25}$	<0.01 *	1.027$_{8,17}$	0.453

Table 5. Density as a function of site ages and based on information collected during point and line transect methods. I report the results of an Fmax test to test whether the variances around mean density estimates are significantly different at $P<0.05$.

Site age (years)	Variance		Fmax$_{(df1\ df2)}$	*P* value (Fmax)
	Point transect	Line transect		
1	0.664	1.010	1.520$_{(2,14)}$	0.418
4	0.279	0.649	2.327 $_{(2,12)}$	0.555
8	0.334	0.416	1.247$_{(2,11)}$	0.680
12	0.383	0.351	0.918$_{(2,14)}$	0.719
16	0.351	0.634	1.803$_{(2,13)}$	0.554
20	0.338	0.401	1.186 $_{(2,10)}$	0.683
24	0.246	0.774	3.138$_{(2,16)}$	0.492
28	0.129	0.304	2.357$_{(2,15)}$	0.746
Unknown	0.173	0.472	2.726$_{(2,17)}$	0.640

DISCUSSION

Blake and Loiselle (2001) indicate that different sampling methods may yield different data and subsequently influence estimates of community variables. I therefore expected that the data obtained from two distance methods would yield different estimates of community variables. My study showed that the communities derived from data collected by the two methods within sites were similar (see Figure 3). The two methods gave similar estimates for species diversity and richness (H' and Margalef's) in the younger regenerating sites (1 to 8 years old) while data from the point transect gave higher estimates in the older regenerating sites and the reference site. Estimates of species richness and diversity in the 20 to 28 year old regenerating sites and the reference site obtained from the point transect method were higher than those based on data collected using line transect surveys. My results agree with those of Shankar Raman and Mudappa (2003) as far as that the point transect performed well in detecting more species than the line transect method. For example, in some of the older sites, the point transect method recorded species such as the Dusky flycatcher (*Muscicapa adusta*), Spotted flycatcher (*Muscicapa striata*), Black-throated wattle eye (*Platysteria peltata*) and Yellow-spotted nicator (*Nicator gularis*), which I did not record during the line transect surveys. A reason for this may be that I had more time to locate and identify species in a single locality during point transects (e.g. Bibby *et al.* 1993, Buckland *et al.* 2001) rather than while continuously moving along a line transect which provided less time to identify and locate bird species within a locality.

Data from the line transect and point transect methods gave similar Shannon diversity and species richness estimates for the youngest site (1 year old). Birds are more conspicuous in open habitats than in dense woodlands (Bibby and Buckland 1987). Therefore increased visibility in younger grasslands probably compensated for less time spent observing during the line transect surveys (Buckland *et al.* 2001). Published work suggests that the point transect may produce both higher (Verner and Ritter 1983) and lower (De Sante 1986) estimates of density than line transects. In contrast Jarvinen (1978), suggests that line transects produce more accurate density estimates than point counts. Verner and Ritter (1983) point out that the area sampled with point counts increases geometrically with distance from the observer, but only linearly with transects. In my study, the point transect produced lower density estimates then the line transect. The differing density estimates could therefore be the result of consistent bias in under estimating distances to birds detected aurally.

An important component of evaluation is to determine how accurately and precisely the methods measure community variables. I could not measure accuracy but inferred precision by evaluating the variance around mean values of estimates. Data from the different methods yielded similar estimates of variance for community variables (H' and Margalef's species richness) for all but two of the sites. Variance estimates derived from mean density estimates from the point transects were not significantly different. Given the lack of statistical significance separating the amount of variance that each method provided I could not conclude that either method is more precise or robust than the other. My assumption that the two survey methods would give different estimates of community variables was only partially incorrect. Whilst density estimates differed throughout all the sites, some of the sites, particularly the older sites, differed significantly for community variables whilst the younger sites tended to be more similar.

I expected that the point transect method would be more efficient than the line transect in its ability to saturate sampling effort within a site due to its perceived ability to detect more species (Bibby *et al*. 1993, Buckland *et al.* 2001). I interpreted efficiency in terms of the time it took to saturate survey effort within a site. I did not incorporate the time that it takes to move between surveying points. Data from the point transect method saturated sampling effort in about half the time of the line transect method. Past research on the efficiency of the two methods differs in their conclusions. For instance, Shankar Raman and Mudappa (2003) hint that the point transect is a more efficient method to use but do not provide any supporting data. Bollinger *et al.* (1988) note that point transects require more time in the field to sample but appreciably less time was on actual surveying. Verner and Ritter (1983) conclude that point and line transects show similar efficiency.

My data indicates that the point transect is more efficient than the line transect. Bibby *et al*. (1993) and Buckland *et al*. (2001) suggest that the point transect allows the observer more time to concentrate on detecting and locating birds instead of concentrating on traversing terrain as in the case of the line transect. My results agree with this in that I was able to detect more species in less time using the point transect survey. In conclusion, my assumption that the point transect method would be more efficient than the line transect method was correct.

CONCLUSION

The survey method used to collect data from which to derive community variables did not influence the estimates of these variables for the younger regenerating sites. However, the point transect method returned higher values for the older than younger rehabilitating sites and a reference site because more species were detected then during line transect surveys. Of importance is that the trends relating to bird communities and their response to rehabilitation efforts, and hence succession, were similar. In terms of the actual time spent surveying the point transect method proved more efficient than the line transect method. I conclude that the point transect method was superior as it detected more species than the line transects in less surveying time than the line transect method.

CHAPTER 4

AN EVALUATION OF BREEDING ACTIVITY BY BIRDS IN REGENERATING COASTAL DUNE FORESTS IN MAPUTALAND

INTRODUCTION

Restoration is associated with habitat change over time. The sensitivity of birds to habitat and vegetation changes makes them good indicators of habitat change (Kritzinger and van Aarde 1998, Canterbury *et al.* 2000, Freshman Browder *et al.* 2002, Perrow and Davy 2002, Diaz *et al.* 2005, Jansen 2005). Studies that use birds as indicators of restoration are usually based on the presence or absence of individual birds or species as detected during formal surveys. An implicit but important assumption of such studies is that the presence of individuals of a species suggests colonization, or even the establishment of a breeding population. However, not all species that are detected in a specific habitat are necessarily breeding there. Some might be transient visitors, either foraging there, or moving through to habitats elsewhere (Owiunji and Plumptre 1998, Blake and Loiselle 2001, Dranzoa 2001). By the same argument, when coastal dune forests are restored, one would like to see the establishment of breeding populations of forest bird species through colonization from adjacent forest habitats. Such breeding activities, in contrast to foraging or dispersal movements, will signify that habitat conditions suitable for breeding to commence are recovering for these species. Overall, breeding reflects on a number of ecological processes underlying the colonization of disturbed areas (Greenwood and Harvey 1982). As mentioned above not all birds that frequent a specific habitat will breed there. Even undisturbed dune forests could thus be expected to host species that have not colonized it. The first important criterion for restoration success is therefore perhaps not whether all observed bird species also breed in restored areas, but whether the bird communities breeding in restored areas are the same as those breeding in undisturbed areas. Second, if a bird is observed as present (recorded in a transect survey) but not breeding in a restored area, but it is present and breeding in a reference area, restoration has not yet been successful.

My first objective for the present study was therefore simply to find out which bird species are breeding in the regenerating sites within the study area. Van Aarde *et al*. (1996), Kritzinger and van Aarde (1998) and Wassenaar *et al*. (2005) monitored the development of the bird assemblages within these areas for 15 years by completing seven surveys for the presence and abundances of bird species in each regenerating site during that period. As part of the monitoring programme they also compared these assemblages to those of undisturbed forests surveyed at the same time. They showed that assemblages in rehabilitating forests (as observed in standard transect surveys) apparently change over time to become more similar to those of undisturbed reference areas (van Aarde *et al*. 1996; Kritzinger and van Aarde 1998; Wassenaar *et al*. 2005). Based on a chronosequence analysis, Wassenaar *et al*. (2005) predicted that it would take between 32 and 38 years after mining to recover the species composition of a forest bird assemblage.

I was particularly interested in whether this apparent recovery in community composition also reflected in the recovery of breeding populations. To determine this, I (with a supporting team) searched for nests in all rehabilitating and reference areas, and surveyed the species assemblage at the same time. I asked the following questions:

1) Are the community compositions and community variables of the observed assemblage and breeding assemblage similar?
2) Does the composition of the breeding assemblage change over time in a similar manner to that of the observed assemblage?
3) Are species that occurred at higher density more likely to be nesting?

I anticipate that the community composition based upon different survey methods will not be similar, as would the change in species richness, diversity and community composition over time for the observed and breeding assemblages. Obviously it is impractical to include breeding surveys in a standard monitoring programme due to time and budgetary constraints. My second objective was therefore to determine to what extent the data from my standard surveys signified breeding (and hence restoration success), and whether standard surveys should perhaps be supplemented by directed searches for nests of specific species in the future. I expect that the presence of bird species will not be sufficient to signify colonisation as deduced from breeding surveys.

MATERIALS AND METHODS

I use the term "nesting assemblage" here to refer to those species for which I found evidence for nesting attempts. This could be either active or old disused nests. I reasoned that if a species has attempted to breed in a site, it is more likely that it will have, or is attempting to establish a breeding population that might be self sustaining. With "observed assemblage", on the other hand, I refer to those species that are recorded during structured point transect surveys. These species are present on the site but there is no information about where they originate from (they may be transient visitors from neighbouring habitats) or whether they do have a viable population locally. I used data collected during point transects (see Chapter 3) to determine bird presence. Bird presence was defined as a recording of a bird using sight or sound within a 60m radius of centre of a point transect. The study site included the eight regenerating sites and the undisturbed reference site (see Chapter 2 for a description of sites). Nest quadrat sampling took place in all of these sites (see Chapter 2). Since point transect sampling was more extensive than line transect sampling I opted only to compare the variables derived from data collected from point transect sampling with that of quadrat sampling.

Quadrat sampling for nests occurred from 24 October 2004 to 8 March 2005. A team of six field workers assisted me. After randomly selected sampling points within a site, we delineated a 30m by 30m quadrat using 120m of rope. Two observers, positioned at opposite corners of the quadrat, spent 30 minutes recording any displays of breeding activities by birds. Thereafter we searched for nests for 15 minutes on each quadrat. We took the presence of eggs, incubation of eggs, and delivery of food to a nest as indications of breeding activity. We used Tarboton (2001) to aid in nest and egg identification where required. Nests that we failed to identify were excluded from the data.

Community variables

I compared community composition among assemblages and among sites using the binary dis-similarity index Ochiai's Distance (OD) (Legendre and Legendre 1998). I used OD because the comparison between two differently sampled assemblages (quadrats and point transects) required an index that only considers the presence or absence of species. OD is the geometric mean of the ratios of the number of shared species to the total number of species in each site:

$$\mathrm{OD} = 1 - \frac{a}{\sqrt{(a+b)(a+c)}}$$

where a is the number of shared species between two sites, and b and c is the number of species unique to the respective sites (Legendre and Legendre 1998). For the species composition evaluation I wanted to determine whether the dis-similarity of the assemblage on each regenerating site to that of the reference site decreased (i.e., whether it became more similar to the reference site assemblage) as the site's age increased. I used the breeding and observed assemblages of the Sokhulu Forest site as the respective reference assemblages for the breeding and observed assemblages of the regenerating sites.

I calculated community structure (the total number of species per site, Shannon diversity (H'), and Margalef's species richness) for both the breeding and observed assemblages using Primer V5 (Clarke and Gorley 2001). These estimates of community structure were then plotted as a function of site age and age related trends subsequently analyzed using the linear regression module in Prism V3.0 (Graphpad Software). Because it is unlikely that an assemblage will recover its community structure linearly (Pimm 1991), I first $\log_{10}$ transformed variables before comparing the linear slopes of the two assemblages (see also Wassenaar *et al.* 2005). An F test was used to determine if the slopes differed significantly from each other and slopes that differed significantly from zero signified regeneration.

I calculated the percentage as well as average number of species detected nesting that were also recorded during the point transect surveys. I also calculated the percentage of species that was not both breeding and observed and similarly evaluated differences among sites. Finally, I calculated the density of each species across all sites and plotted this against the number of nests that were found for each species, across the whole study area.

RESULTS

It took 288 hours to complete 437 nest quadrat surveys – these yielded 214 observations of breeding activities for 36 species. The 292 point transects yielded 2145 observations of 62 species (see Appendix 3).

Community composition

Table 6 shows us that across 9 sites the breeding assemblage was a subset of the observed assemblage. Nearly 84% of the breeding assemblage detected in the 9 sites were also observed during point transect surveys. However, the two assemblages shared a small percentage of species indicating that a large portion of the observed assemblage was not observed breeding. A list of the species observed during point transect and nesting surveys is presented in Appendix 4. Species typical of early successional stages (1-4 years) were Rattling cisticola (*Cisticola chenia*), Green-backed camaroptera (*Camaroptera brachyuran*), Grassveld pipit (*Anthus cinnamomeus*) and Red-backed mannikin (*Spermestes bicolor*). While these species were abundant in the earlier stages of succession, I found no evidence of them breeding. The intermediate regenerating sites (8-20 years) were dominated by species such as Tawny-flanked prinia (*Prinia subflava*), Black-eyed bulbul (*Pycnonotus barbatus*), Green-backed camaroptera and Yellow-bellied bulbul (*Chlorocichla*

Table 6. The number of bird species detected within sites during point transect and quadrat surveys for nests. The numbers of nesting species recorded during point transect surveys and the percentage species shared by the surveys on each site is provided. The percentage shared species was obtained by subtracting the species detected in nest quadrats but not observed from the total observed species and then calculating the shared species as a percentage of the total number of species per site for both methods.

Site	Site age (years)	Area (ha)	Number of species		Number (% in brackets) of species in the nesting surveys that were also recorded during point transect surveys	Percentage shared species
			Observed	Breeding		
Post mining rehabilitating sites						
Site 8	1	39	17	1	1(100)	5.88
Site 7	4	32	19	0	0(0)	0.00
Site 6	8	104	16	2	2(100)	12.50
Site 5	12	87	16	10	8(87.5)	44.44
Site 4	16	124	15	7	6(93.3)	37.50
Site 3	20	80	18	8	7(94.4)	36.84
Site 2	24	190	24	9	9(100.0)	37.50
Site 1	28	76	34	13	11(94.1)	30.56
Reference site						
Sokhulu	Unknown	500	39	23	19(89.7)	44.19

flaviventris), all of which were recorded breeding during the nest quadrat surveys on these sites. Surprisingly, I also recorded Thick-billed weaver (*Amblyospiza albifrons*), Yellow weaver (*Ploceus subaureus*), Blue-mantled crested flycatcher (*Trochocercus cyanomelas*) and Woodward's batis (*Batis fratru*m) during point transect surveys. According to their breeding habits (Tarboton 2001) these species would not normally occur within these sites. The older sites (24-28 years) and reference site were dominated by Forest weaver (*Ploceus bicolor*), Green-backed camaroptera, White-eared barbet (*Stactolaema leucotis*) and Golden-rumped tinker bird (*Pogoniulus bilineatus*), all of which have also been noted to breed on these sites. I did not note breeding activities for Trumpeter hornbill (*Bycanistes bucinator*), Livingstone's turaco (*Tauraco livingstonii*), Black-throated wattle eye and Woodward's batis.

The Ochiai's distance ($\log_{10}$) to the reference site for both assemblages decreased significantly (Observed assemblages $F_{[1,6]}$=15.24, *P*=0.008, breeding assemblage $F_{[1,6]}$=13.45, *P*=0.01) with increasing age. The rates at which these assemblages approached the reference assemblage (the slopes of the regressions) were similar ($F_{[1,12]}$=0.01, *P*=0.89). The similarity in the Y-intercepts of the two assemblages ($F_{[1,13]}$=2.15, *P*=0.17) suggest that these communities were equally dis-similar to their respective reference sites at the onset of regeneration (Figure 5).

Community variables

The number of species ($\log_{10}$) in the assemblages constructed from data obtained through the different surveys increased significantly with site age (Observed assemblages $F_{1,7}$=14.03, *P*=0.007, breeding assemblages $F_{1,7}$=28.51, *P*=0.001). The difference in the rate of increase of the observed and breeding assemblages was significant ($F_{1,14}$=12.1, *P*=0.003) (Figure 6) with the breeding assemblage increased at a faster rate, possibly due to the few nests recorded on sites less than eight years old.

Species diversity ($\log_{10}$) increased significantly with site age for both the observed and breeding assemblages (Observed assemblage $F_{1,7}$=16.33, *P*<0.001, breeding assemblages

$F_{1,7}=21.25$, $P<0.001$). In addition, the rate at which the diversity estimates for the two assemblages increased was similar ($F_{1,14}=0.96$, $P=0.344$). The difference in Y-intercepts was significant ($F_{1,15}=16.34$, $P<0.001$) (Figure 7) due to the few species found breeding in the regenerating sites less than eight years old.

Correlation of nest and species specific densities

A regression of the species specific number of nests against species specific densities per hectare suggest that the number of breeding individuals increased significantly with increasing density ($F_{1,8}=25.79$, $P<0.001$) (Figure 8). However, only 44% of the variability in nest densities is explained by densities derived from point transect surveys.

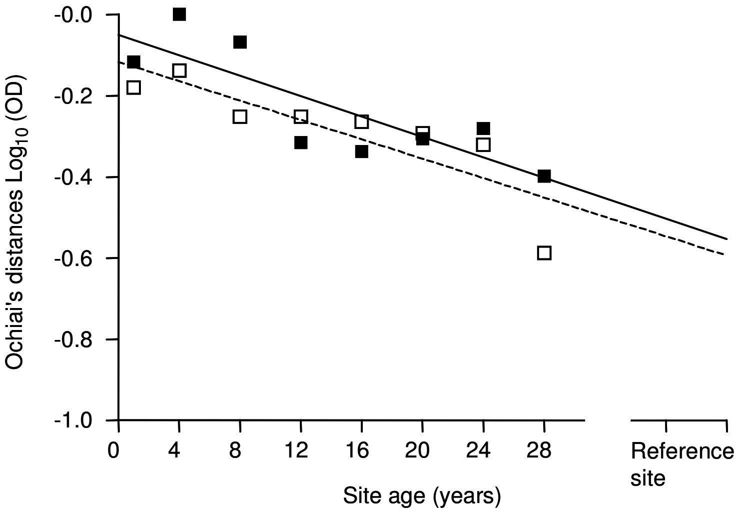

Figure 5. Ochiai's Distance ($\log_{10}$) as a function of site age based on two separate data sets. The closed squares and solid line represent observed assemblages. The open squares and dashed line represent the breeding assemblages.

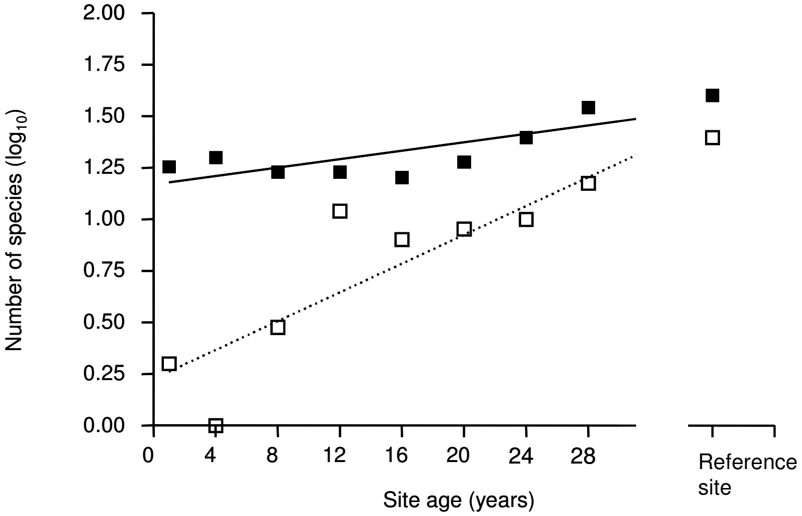

Figure 6. The number of species ($\log_{10}$) as a function of site age for two data sets. The solid line and closed squares represents assemblages based on point transect surveys while the dashed line and open squares represents assemblages based upon nest quadrat surveys. I also present the number of species recorded in the reference site.

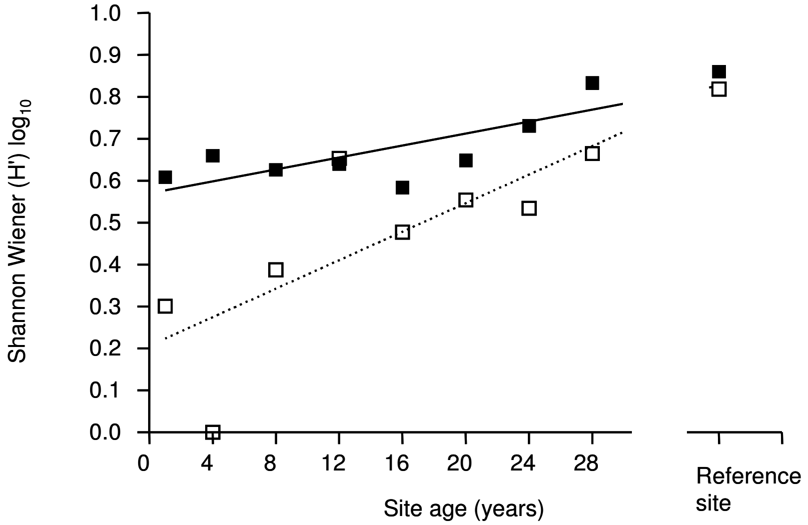

Figure 7. Shannon diversity (H') estimates ($\log_{10}$) as a function of regeneration age for two data sets. A solid line and closed squares represent observed assemblages. A dashed line and open squares represent assemblages detected by nest quadrat surveys. The estimates for the reference site are included.

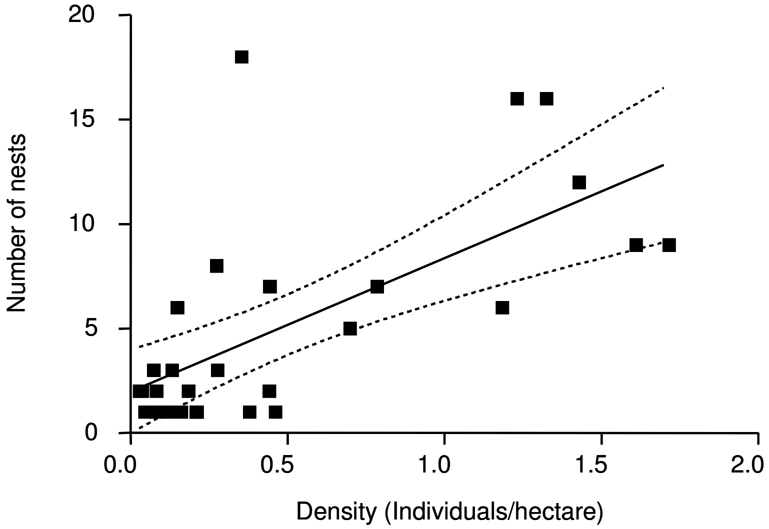

Figure 8. The number of nests as a function of density for each of the species that nested in the study area. Dotted lines indicate 95% confidence intervals.

DISCUSSION

Earlier studies suggest that the presence of a bird species is an unreliable indicator of colonisation (Blake and Loiselle 2001, Dranzoa 2001). In spite of this a number of studies assumed that presence indicates establishment (Canterbury *et al*. 2000, Freshman Browder *et al*. 2002, Perrow and Davy 2002). This assumption may be flawed. Certainly, in my study, I detected species, which according to their breeding habits (Tarboton 2001) would not be breeding within habitats they were detected within.

Based upon arguments that presence does not equate to colonisation (Owiunji and Plumptre 1998, Dranzoa 2001), I anticipated that the breeding and observed assemblages within my study area would provide different community compositions. I did detect species such as Woodward's batis, Thick-billed weaver and Village weavers foraging in sites where they would not be expected to be breeding i.e. the preferred habitat for Thick-billed weavers to breed in is reed beds and not coastal dune forest. The presence of these species would certainly have added to the dissimilarity in the community compositions derived from transect and breeding surveys. However, most species noted breeding were also recorded during point transect surveys and those species mentioned above may be exceptions. In as much as I detected species that would not be breeding, I also suspect that I did not detect all species that were breeding within my study sites. For instance, the Trumpeter hornbill, Livingstone's turaco and Purple-crested turaco should at least have been detected breeding in the reference sites, but were not. After all earlier workers in the area (Kritzinger 1996, Niemand 2001), like me, noted these species in most patches of intact forest in the region. I presume that the cryptic nature of their nests precluded detection during breeding surveys and not because they were not breeding in the area. Forest specialists that are limited to coastal dune forest and that occurred at low densities were also not recorded breeding. These included Black-throated wattle eye and Woodward's batis, which would certainly not be breeding elsewhere. This may simply be due to the rarity of their nests, a notion supported by the apparent relationship between bird and nest densities (see Figure 8).

Nearly 84% of the breeding assemblage detected in the 9 sites were also observed during point transect surveys. Most of the birds seen during point transect surveys apparently did not nest as suggested by the breeding surveys (see Table 6). Should those species discussed above also be breeding here, then the composition of the assemblages reflected by the nesting and point transect surveys would be even more similar. However, this was not the case with the breeding and observed assemblages on average across all sites 27.65% of their species. The breeding assemblage shares 84% of its species with the observed assemblage meaning that it is a subset of the observed assemblage. My expectation that the two community compositions would be different was therefore correct as the breeding assemblage was more of a subset than a similar community.

I expected that age related trends for species richness and diversity derived from the survey methods would not be similar. I found an increase in bird species diversity and species richness with increasing regeneration age for both assemblages. In terms of species richness and diversity, the breeding assemblages increased in similarity to a reference site at a faster rate than the observed assemblage. The loss of structural components of old growth forests, such as large canopy trees, affects the populations of species that depend on these elements (Berg *et al.* 1994). Several studies suggest that forest birds may depend on forest structural elements, particularly understorey birds and large tree users (see Diaz *et al*. 2004). For instance, I first recorded the White-eared barbet and the Golden-rumped tinker bird, species with specialised nesting requirements, in older regenerating sites (24-28 years old). These habitat specialist species are particularly important – the more demanding the species' requirements, the more its breeding presence signifies restoration success (Karr 1981). The presence of these species possibly indicate that resources such as nesting opportunities are becoming available to different species as site age increases. Kikkawa (1982) found a correlation between high bird species richness and diversity and structurally complex habitats. According to Kritzinger (1996), the structural complexity of the coastal dune forest increases with regeneration age. This increase in structural complexity within the regenerating sites may therefore explain the increase in bird species richness and diversity within regenerating sites. The few species that breed in the relatively young regenerating sites (<8 years old) may explain the apparent high rate of regeneration implied by the nesting surveys. I attributed this to birds either not breeding in the earlier sites or to my failure to locate their nests. The progress of the observed and breeding assemblages towards a reference site in my study agree with other studies of successional development in bird communities (e.g. Smith and MacMahon 1981, May 1984, Monkkonen and Helle 1987) as well as those of Kritzinger and van Aarde (1998). I conclude that age related trends derived from nest quadrat surveys and point transect sampling for species richness and diversity are similar and that my initial assumption that they would be significantly different to be incorrect.

I anticipated that the presence of a bird species during distance sampling would not be sufficient to signify colonisation. Results from my study show that both assemblages increased in similarity to the assemblages of a reference site in a similar manner. Wassenaar *et al.* (2005) use line transect data to measure the rate at which various taxa approached an undisturbed reference site in terms of similarity of community composition. Their findings indicate that the bird assemblage approached the reference site in similarity as site age increases. My results concur with theirs but more importantly, the breeding assemblage progressed towards the reference site in a similar manner as the observed assemblage. My initial expectations proved to be incorrect and I conclude that the presence of a bird species during distance sampling is a reliable indicator of colonisation.

CONCLUSION

While I did detect bird species unlikely to be nesting within regenerating sites I did establish that the breeding assemblage is a subset of the observed population. Age related trends for species richness and diversity shown by the observed and breeding assemblages were similar. Despite my inability to detect certain species the breeding assemblage still approached the reference site in similar manner to the observed assemblage. I conclude that the presence of bird species within a site is sufficient to deduce that colonisation did take place.

CHAPTER 5

SYNTHESIS

Birds are sensitive to changes in habitat structure and composition and their presence and/or absence reflects on ecological change (Canterbury *et al.* 2000, Freshman Browder *et al.* 2002, Perrow and Davy 2002, Wassenaar *et al.* 2005). Whilst bird communities feature strongly in restoration studies, the methods used to collect data may influence inferences. After all researchers can only make inferences from their findings if the methods of data collection return reliable information. I focused on evaluating the influence of data, collected using three different sampling methods, on bird community composition and structure. In Chapter 3, I compared the community compositions and estimates of community variables derived from data collected by two distance methods to establish which of these methods may be preferable for use in my study area. In Chapter 4, I compared two assemblages, one based upon point transect data and the other based upon breeding records, to determine if the presence of birds signified colonisation. Here, I review my research findings and the implications these may have for future work on bird communities within the study area.

The data provided by the line transect and point transect surveys yielded similar community variables for the younger regenerating sites (<8 years old), but the point transect method gave higher estimates for the older regenerating sites. The point transect and line transect methods yielded significantly similar variance estimates (Table 5). In terms of the amount of time spent surveying the point transect method proved more efficient. The point transect method may therefore be considered as superior to the line transect method and this should be considered for further work within the study area. The point transect method, although apparently suited to environmental conditions within my study area may not be the most suitable method to use elsewhere. For instance, Anderson and Ohmart (1981) indicate that the line transect method was more efficient if stands of vegetation are large enough to establish transects of 700-800m in length and provided that the topography allows unhindered movement. This was not the case in my study area where line transects were limited to a length of 300m and topography hampered movement. Researchers should therefore take into account the dimensions of their study area when planning a survey.

While distance methods provide a robust manner in which to survey bird communities (Buckland *et al.* 2001) one should not view them in isolation. Methods involving sampling by sight and sound, such as the distance methods, have the limitation of under representing species that are cryptic or less active and those which sing infrequently (Ntiamoa-Baidu *et al*. 2000). There is agreement that 1) no single method adequately samples all (or even most) bird species in complex habitats (Blake 1992, Pomeroy 1992), 2) that a combination of methods is most likely to provide a more complete enumeration of species in a given area (Loiselle and Blake 1992, Rappole *et al.* 1993) and 3) the choice of method should be determined by the objective of the study (Pomeroy 1992). This raises the question as to whether a combination of methods should possibly be considered for future use in the study area.

Spot mapping is an effective method used to produce maps of the distribution of birds within a study area (Bibby *et al*. 1993). A major disadvantage of spot mapping is that it is extremely time consuming as well as unsuitable for large areas (Bibby *et al.* 1993). Karr (1981) argues that mist netting offers the best procedure for censusing birds on the basis that they avoid the bias of inadequate knowledge of resident avifauna and provide a random, unbiased sample of birds moving in the space sampled. He does however admit that mist netting is only adequate for under storey birds. These methods would therefore both be unsuitable for future use in the study area. One recently developed approach to the rapid assessment of avifauna, the MacKinnon list method (see Mackinnon and Phillips 1993), has been increasingly adopted worldwide (Poulsen *et al.* 1997, Fjeldsa 1999, Trainor 2002). This methods involves all species seen and heard being grouped into consecutive lists of equal length. A species accumulation curve is generated from adding those species not recorded on any previous list to the total species number. 10-species lists (Herzog *et al.*

2002) represent the best compromise between stable richness estimation curves and robust sample size. The advantages claimed for the technique relative to the distance methods include time efficiency and relatively greater observer independence; the entire available time period is used to collect data and hence there is far less data loss than in other survey methods (Fjeldsa 1999). Poulsen *et al*. (1997) claim that the method is suitable for judging 1) when a site is adequately surveyed, 2) species richness, 3) the relative abundance of each species and 4) an α- index of diversity. O' Dea *et al*. (2004) argue in favour of collecting additional data to verify those obtained during systematic surveys. These may then be used to generate a comparative index of sampling effort and improve the assessment of the species richness within an area. In line with this reasoning as well as the results obtained in this study, I propose that the Mackinnon list method be considered for future research within the study area along with point transects.

In Chapter 4, I examined the variables of bird assemblages inferred from data collected during point transect surveys and breeding surveys for the eight regenerating sites and reference site. Bird presence does not equate to colonisation, which is a central issue in restoration success. Their presence therefore may be misleading and thus raises the question as to whether bird presence data are sufficient to infer successional development in response to a rehabilitation programme. In my study, the presence of non-breeding species within regenerating sites reinforced the notion that not all observed species would be breeding (Dranzoa 2001). The number of species and species diversity also increased with site age for both observed and breeding assemblages. In addition, both observed and breeding assemblages approached the assemblages of an undisturbed reference site in similarity as site age increased. These results suggest that dispersal from natal areas is taking place, that it is occurring in sufficient numbers to result in mating and breeding, and that specific habitat resources for nesting are present. Such activities, in contrast to foraging or transitory movements, signify that habitat conditions are favouring the colonization of regenerating sites by birds typical of the region. Most important though is the return of species which require specialised breeding sites such as hole nesters. Wassenaar *et al.* (2005), in their study of community convergence, remarked that complete convergence may depend on a subset of species, such as hole nesters, that require a particular habitat component such as large old trees for nesting holes. According to Dranzoa (2001) species with special breeding requirements, such as hole nesters, are often the last to colonise areas due to a lack of suitable nest sites. The apparent recovery of these specific communities further reinforces the conclusions of Wassenaar *et al.* (2005) that bird communities (based on presence data) of regenerating sites are developing towards those of undisturbed reference sites. Whilst these results are encouraging I must take into account arguments put forward that communities are not static over time but are dynamic (Bengtsson *et al.* 1997). Throughout my study I made the assumption that the bird community of the reference site would represent the end state of the regenerating sites. In a study of bird communities in British woodlands they show that variability within communities increases with time (Bengtsson *et al.* 1997) i.e. the composition of a bird community may change annually. The usefulness of indicator communities for detecting change would therefore be limited if communities varied over time in absences of changes in the environment. Wassenaar *et al.* (2005) considered this and concluded that, for various reasons, the number of possible end states for the coastal regenerating sites was small. By assuming that these spatially separated sites are developing along similar pathways over time and that the number of possible end states are small (Wassenaar *et al.* 2005), I can assume that my conclusions are in fact valid.

My results therefore have implications for studies using bird communities as surrogates of succession. I have shown that the presence of species is sufficient to signify colonisation when compared to the progress of breeding assemblages. This is important as it adds confidence to the previous conclusions relating to succession within my study area as well for other studies based upon data derived from surveys of bird presence alone. In addition, the fact that presence signifies colonisation negates the need to conduct time consuming nesting surveys.

Whilst the recovery of bird communities implies that they are responding to the rehabilitation programme implemented by Richards Bay Minerals, these results may not however mean that the programme has been successful. According to Franklin *et al*. (1981) ecosystems are comprised of three primary attributes namely composition, structure and function. Noss (1990) elaborates on Franklin's concept and suggests indicator variables for monitoring and assessing terrestrial biodiversity at four levels of organization namely regional landscapes, community-ecosystems, population-species and at a genetic level nested within Franklin's definition. Here, I have only evaluated the progress of bird communities at a community level which forms a small part of what would be required to monitor an entire system such as the one in my study area. In order to determine fully whether the successful restoration of the coastal dune forests has been achieved one would need to evaluate the progress of other taxa as well as consider various ecological processes and how they influence the structure and functioning of the emerging ecosystem. Whilst bird communities are good indicators of habitat change, and subsequently succession, other factors need to be considered to evaluate fully whether the rehabilitation of the coastal dune forests has in actual fact been successful.

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SUMMARY

The post mining rehabilitation of coastal dune forests by Richards Bay Minerals north of Richards Bay in Maputaland has given rise to a sere of known age regenerating sites. This unique setting has provided scientists with an outdoor laboratory in which to study the mechanics of coastal dune regeneration. In this study, I have focused on two aspects of birds and rapid assessment techniques. Firstly, I evaluated the performance of two distance methods in their ability to estimate the community variables of bird communities as well as to saturate sampling effort within a site. Eight regenerating sites as well as a reference site were surveyed from 1 January to the 15 March 2005 using the line and point transect methods. I used Primer V5 to calculate the total number of species, Shannon Wiener diversity index (H') and Margalef's species richness for each method within a site. I used DISTANCE 4.1 to estimate the relative abundance of each species. Data collected provided similar estimates of community variables in the younger sites. The point transect method provided higher estimates in the older regenerating sites as well as the reference site. The trends relating to bird communities and their response to rehabilitation efforts, and hence succession, were similar and were characterised by changes in species typical of grasslands to communities more typical of coastal dune forests. There was no significant difference between variance estimates generated by the two methods. The point transect method proved to be more efficient in terms of time spent surveying. My conclusion was that the point transect method was a possibly a more superior method than the line transect method to use within the study area based on the lower amount of time required to saturate sampling using this method.

Secondly, I compared two bird communities, one based upon point transect data and the other upon breeding records to determine if bird colonization has occurred. Community structure (the total number of species per area, Shannon Diversity (H'), and Margalef's species richness index) were calculated using Primer V. I compared community composition between assemblages

and amongst sites using the binary dis-similarity index, Ochiai's Distance (OD). The breeding assemblage was a subset of the observed assemblage. The breeding assemblage approached the reference site at a similar rate as the observed assemblage in terms of species composition. I concluded that the presence of bird species within a site is sufficient to deduce that colonisation did take place. Given that presence is a reliable indicator of colonisation This negates the need to undertake nesting surveys.

In Chapter 5, I synthesised my findings. I proposed that the point transect method, be combined with an alternative rapid assessment technique, the Mackinnon list method, in order to improve the efficiency of sampling within the study area. I also examined the results of the nest quadrat survey and how they relate to the restoration programme at Richards Bay Minerals in terms of the success of the programme.

OPSOMMING

Die rehabilitasie van gemynde kusduinewoud deur Richards Bay Minerals in Maputaland, noord van Richardsbaai, het aanleiding gegee tot ‘n reeks van gebiede met bekende ouderdomme in ‘n proses van regenerasie. Hierdie reeks gebiede het aan wetenskaplikes die geleentheid geskep om die herstel van kusduinewoud te bestudeer. In die studie het ek op twee aspekte gekonsentreer, naamlik voëls en die twee digtheidsbepaling tegnieke. Ek het begin deur twee afstandsmetodes te vergelyk en hulle potensiaal om veranderlikes in voël gemeenskappe te bepaal sowel as om vas te stel wanneer die versadigingspunt vir data insameling vir beide bereik word. Die studie is gedoen in die tydperk Januarie tot 15 Maart 2005. Ek het ‘n voël opname in agt gebiede, in verskillende tye van regenerasie en ‘n verwysingsgebied, gedoen deur gebruik te maak van ‘n lyntransek en punttransek metode. Ek het die Primer V5 program gebruik om die totale hoeveelheid spesies binne ‘n gebied te bepaal, sowel as die Shannon Wiener diversiteitsindeks (H’) en Margalef se spesie rykheid indeks. Ek het die DISTANCE 4.1 program gebruik om die relatiewe hoeveelheid individue vir elke spesie te bepaal. Die data versamel in die jonger gebiede het dieselfde gemeenskapsveranderlikes gehad vir albei metodes.Die punt-transek metode het hoër skattings in beide die ouer geregenereerde gebiede as in die verwysingsgebied gehad. Die neiging van die voëls wat in ‘n gebied voorkom, en hulle reaksie op die rehabilitasie van die verskillende gebiede en gevolglike suksessie, was dieselfde, en is gekarakteriseer deur verandering in spesies wat in graslande verwag word na gemeenskappe wat meer met kusduinewoud geassosiëer word. Die lyntransek metode het groter variasie getoon rondom die gemiddelde digtheidsskattings wat aandui dat die punttransek metode minder vatbaar is vir veranderlikes. Die punttransek metode blyk om meer tydeffektief te wees. En die gevolgtrekking is dus dat die punttransek metode ‘n beter metode is as die lyntransek metode vir gebruik in hierdie studie gebied.

Tweedens het ek twee voël gemeenskappe met mekaar vergelyk, een gebasseer op die punttransek inligting, en die ander op die nes rekords om vas te stel of kolonisasie plaasgevind het. Die Primer V program is gebruik om die gemeenskapsstruktuur (die totale hoeveelheid spesies per gebied, Shannon Diversity (H'), en Margalef se spesie- rykheidsindeks) te bereken. Ek het die gemeenskapstruktuur tussen gebiede en die samevoeging van die twee stelle data vergelyk deur van die binêre disimilêre indeks gebruik te maak, "Ochiai's Distance (OD)". Die gemeenskappe wat neste in die gebied het, en gevolglik in die gebied broei, is 'n gedeelte van die totale voëls aangeteken in die gebied. Die gemeenskap wat in 'n gebied broei, en die gemeenskap wat in die verwysings gebied gesien is, nader die verwysingsgebied met dieselfde tempo. My gevolgtrekking is: die voorkoms van voëls in 'n gebied is genoegsame bewys van kolonisasie.

In hoofstuk 5 som ek my bevindings op. Ek stel voor dat die punttransek metode, die metode van voorkeur, gekombineer moet word met 'n alternatiewe metode, naamlik die Mackinnon lys-metode om effektiwiteit van versameling van inligting vir die gebied te verbeter. Ek het ook die inligting van nes voorkoms met die restorasie program van Richards Bay Minerals vergelyk in terme van die sukses van die program.

Appendix 1. A description of the vegetation of the post mining regenerating sites, undisturbed reference site and the ecostrip that were surveyed as part of the present study (Niemand 2001)

Regeneration year	Age (yrs)	Hectares (ha)	Vegetation description
2001-2004	1 to 4	39	Secondary grassland with Acacia kosiensis shrubs scattered throughout the site. Grasses include *Dactyloctenium germinatum*, *Pennisetum americanum*, *Panicum maximum*, Sorghum spp., *Crotalaria juncea* and herbs such as *Neonotonia wightii*, *Glorosia superba* and *Canavalia rosei*.
1997-2000	5 to 8	32	Acacia kosiensis dominated shrubland of about 3 meters high with various grass species such as *Digitaria diversinervis* and *Dactyloctenium australe*. Broadleaf species include *Carrisa bispinosa*, *Passerina rigida* and *Chrysanthemoides monolifera*.
1993-1996	9 to 11	104	*Acacia kosiensis* dominated thicket of approximately 9 meters with sparse undergrowth. Broadleaf species include *Vepris lanceolata* and *Brachylaena discolor*, all characteristic of coastal dune scrub. Undergrowth consists primarily of herbs such as *Drimiopsis maculata, Zehneria scabra* and *Crommelina spp.*
1990-1992	12 to 14	87	*Acacia kosiensis* dominated woodland of approximately 12 -14 meters in height. *Brachylena discolor*, *Kraussia floribunda*, *Canthium inerme*, *Vepris lanceolata*, *Scutia myrtina* and *Rhus nebulosa* are the most dominant broadleaf species of this site. The undergrowth consists of *Dactyloctenium australe*. *Laportea pendacularis* and *Asystasia gangetica*
1988-1990	15 to 17	124	Acacia kosiensis dominated woodland with a canopy 17-18 meters high. Other woody species include *Mimusops caffra*, *Celtis africana*, *Treme orientalis*, *Brachylaena discolor*, *Apodytes dimidiata*, *Vepris lanceolata* and *Trichelia emetica*. The groundcover consists mostly of *Digitaria diversinervis*, *Laportea penducularis* and *Achyranthes aspera*.

1985-1987	18 to 20	80	*Acacia kosiensis* dominated woodland of 19-20 meters tall. The woody species are similar to the 15-18 year old site. Other woody species include *Carissa macrocarpa*, *Draceana aletriformis*, *Zanthoxylum capensis*, *Allophylus natalensis, Deinbolla oblongifolia*, *Sideroxylon inerme* and *Psychotria capensis*. The groundcover consists of *Laportea penducularis*, *Asystasia gangetica*, *Digitaria diversinervis*, *Pupalia lappacea*, *Senecio deltoideus* and *Secamone filiformis.*
1981-1984	21 to 24	190	*Acacia kosiensis* dominated woodland of about 21-24 meters tall. Broadleaf species include *Teclea gerrardi*, *Xanthoxylum capense*, *Allophylus natalensis*, *Canthium inerme* and *Maytenus nemorosa*. Other broadleaf species dominate in an emerging midstratum layer including *Tricalysia sonderiana*, *Kraussia floribunda* and *Strelitzia nicolai*. The undergrowth consist of shrubs such as *Achyrathes aspera*, *Asystasia gangetica* and *Laportea peduncularis*. Grass species include *Digitaria diversinervis*, *Dactyloctenium australe* and *Panicum maximum*.
1978-1980	25 to 28	76	Acacia dominated woodland. Woody species include Canthium inerme, Mimusops caffra, Allophylus natalensis, Celtis africana. Midstratum includes Dracaena aletriformis, Kraussia floribunda, Peddia africana, Phoenix reclinata. Shrubs include Brachylaena discolor, Rhus nebulosa and Rhus natalansis. Herbs include Laportea peduncularis, Asystasia gangetica and Achyranthes aspera. Digitaria diversinervis, Dactyloctenium australe and Panicum maximum.
Sokhulu forest		500	An unmined forest (reference site) dominated by woody species such as *Mimuspos caffra*, *Celtis africana* and *Ziziphus mucronata* of over 30 meters forming the canopy. Other woody species include *Croton sylvaticus*, *Gymnosporia mossambicensis*, *Chaetacme aristata*, *Englerophytum natalense*, *Cordia caffra*, *Dryptes natalensis*, *Ochna arborea*, *Diospyros natalensis*, *Erythroxylum emarginatum*, *Strychnos gerrardi*, *Teclea gerrardi*, *Vangueria randi*, *Clausena anisata*, *Chionanthus peglearae* and *Trichilia dregeana*. The understorey consists mostly of *Asparagus falcatus*, *Isoglossa woodii*, *Scadoxus puniceus*, *Tragia glabratta*, *Pyrenacantha grandiflora* and *Cyanchum ellipticum*.

Ecostrip	Unknown	Plant species consist mostly of salt spray tolerant species. These include tree species such as *Strelitzia nicolai*, *Mimusops caffra*, *Brachylaena discolor* and *Allophylus natalensis*.
Casuarina plantations	1235	Entirely Casuarina equisetifolia dominated
Gum plantations	Unknown	Timber species *Eucalyptus grandis* and the exotic weeds *Lantana camara* and *Psidium guajava* dominate. Indigenous tree species that manage to survive in this locality are *Annona senegalensis*, *Tricalysia sonderiana* and *Kraussia floribunda*.

Appendix 2. List of all bird species that have been recorded within the study area during sampling for the Ecological Monitoring Programme (unpublished CERU database) and by Kritzinger (1998) and Niemand (2001).

Family	Common Name	Scientific Name
Anatidae	Spur-winged goose	*Plectopterus gambensis*
Threskiornithidae	Hadeda ibis	*Bostrychia hagedash*
Accipitridae	Cuckoo hawk	*Aviceda cuculoides*
	Long-crested eagle	*Lophaetus occipitalis*
	Southern banded snake eagle	*Circaetus fasciolatus*
	Brown snake eagle	*Circaetus cinereus*
	African fish eagle	*Haliaeetus vocifer*
	Steppe buzzard	*Buteo buteo*
	Crowned eagle	*Stephanoaetus coronatus*
	African goshawk	*Accipiter tachiro*
	Little sparrowhawk	*Accipiter minullus*
	African marsh harrier	*Circus ranivorus*
	African harrier hawk	*Polyboroides typus*
	Honey buzzard	*Pernis apivorus*
Rallidae	Buff-spotted flufftail	*Sarothrura elegans*
Columbidae	Red-eyed dove	*Streptopelia semitorquata*
	Tambourine dove	*Turtur tympanistria*
	Lemon dove	*Aplopelia larvata*
	Green pigeon	*Treron calva*
	Emerald-spotted wood dove	*Turtur chalcospilos*
Musophagidae	Livingstone's turaco	*Tauraco livingstonii*
	Purple-crested turaco	*Tauraco porphyreolophus*
Cucullidae	Emerald cuckoo	*Chrysococcyx cupreus*
	Klaas's cuckoo	*Chrysococcyx klaas*
	Diederik cuckoo	*Chrysococcyx caprius*
	Red-chested Cuckoo	*Cuculus solitarius*
	Green coucal	*Ceuthmochares aereus*
	Burchell's coucal	*Cenrtopus burchellii*
Tytonidae	Barn owl	*Tyto alba*
Strigidae	Marsh owl	*Asio capensis*
Caprimulgidae	European nightjar	*Caprimulgus europaeus*
Colligidae	Red-faced mousebird	*Urocolius indicus*
	Speckled mousebird	*Colius striatus*
Trogonidae	Narina trogon	*Apaloderma narina*

Family	Common Name	Scientific Name
Coracidae	European roller	*Coracias garrulus*
	Lilac-breasted roller	*Coracias caudata*
Halyconidae	Pygmy kingfisher	*Ispidina picta*
	Brown-hooded kingfisher	*Halcyon albiventris*
Meropidae	Little bee eater	*Merops pusillus*
Buceropitidae	Trumpeter hornbill	*Bycanistes bucinator*
	Crowned hornbill	*Tockus alboterminatus*
Capitonidae	White-eared barbet	*Stactolaema leucotis*
	Black-collared barbet	*Lybius torquatus*
	Red-fronted tinker barbet	*Pogoniulus pusillus*
	Golden-rumped tinker barbet	*Pogoniulus bilineatus*
	Crested barbet	*Trachyphonus vaillantii*
Indicatoridae	Scaly-throated honeyguide	*Indicator variegatus*
Picadae	Golden-tailed woodpecker	*Campethera abingoni*
	Cardinal woodpecker	*Dendropicos fuscescens*
Alaudidae	Rufous-naped lark	*Mirafra africana*
Hirondidae	Black saw-wing	*Psalidoprocne holomelas*
	Lesser-striped swallow	*Hirundo abyssinica*
	Barn swallow	*Hirundo rustica*
Campahagidae	Black cuckooshrike	*Campehaga flava*
	Grey cuckooshrike	*Coracina caesia*
Dicruridae	Fork-tailed drongo	*Dicrurus adsimilis*
	Square-tailed drongo	*Dicrurus ludwigii*
Oriolidae	Black-headed oriole	*Oriolus larvatus*
	European golden oriole	*Oriolus oriolus*
Corvidae	Pied crow	*Corvus albus*
Pycnonotidae	Dark capped bulbul	*Pycnonotus barbatus*
	Terrestrial brownbul	*Phyllastrephus terestris*
	Sombre greenbul	*Andropadus importunus*
	Yellow-bellied greenbul	*Chlorocichla flaviventris*
	Eastern nicator	*Nicator gularis*
Turdidae	Kurrichane thrush	*Turdus libonyana*
	Spotted ground thrush	*Zoothera guttata*
	Stonechat	*Saxicola torquata*
	Red-capped robin thrush	*Gossypha natalensis*
	Cape robin chat	*Cossypha caffra*
	White starred robin	*Pogonocichla stellata*
	Cape rock thrush	*Montocola rupestris*
	Bearded scrub robin	*Erythropygia quadrivirgata*

Family	Common Name	Scientific Name
	White browed robin	*Erythropygia leucophrys*
	Brown scrub robin	*Erythropygia signata*
Sylividae	Marsh warbler	*Acrocephalus palustris*
	Willow warbler	*Phylloscopus trochilus*
	Garden warbler	*Sylvia borin*
	Icterine warbler	*Hippolias icterina*
	Yellow-breasted apalis	*Apalis flavida*
	Rudd's apalis	*Apalis ruddi*
	Long-billed crombec	*Sylvietta rufescens*
	Green-backed camaroptera	*Camaroptera brachyura*
	Zitting cisticola	*Cisticola juncidis*
	Pale-crowned cisticola	*Cisticola brunnescens*
	Croaking cisticola	*Cisticola natalensis*
	Rattling cisticola	*Cisticola cheniana*
	Tawny-flanked prinia	*Prinia subflava*
Muscicapadae	Dusky flycatcher	*Muscicapa adusta*
	Spotted flycatcher	*Muscicapa striata*
	Ashy flycatcher	*Muscicapa caerulescens*
	Black flycatcher	*Melaenornis pammelaina*
	Pallid flycatcher	*Melaenornis pallidus*
	Woodwards' batis	*Batis fratrum*
	Black-throated wattle-eye	*Platysteira peltata*
	Blue-mantled crested flycatcher	*Trochocercus cyanomelas*
	Paradise flycatcher	*Terpsiphone viridis*
	Grey-tit flycatcher	*Myioparus plumbeus*
Motacillidae	Grassveld pipit	*Anthus cinnamomeus*
	Cape wagtail	*Motacilla capensis*
	African pied wagtail	*Motacilla aguimp*
	Yellow-throated longclaw	*Macronyx croceus*
Lanidae	Common fiscal	*Lanius collaris*
	Red-backed shrike	*Lanius collurio*
	Lesser grey shrike	*Lanius minor*
Malacanotidae	Southern boubou	*Laniarius ferrugineus*
	Black-backed puffback	*Dryoscopus cubla*
	Gorgeous bush shrike	*Telephorus quadricolor*
	Orange-breasted bush shrike	*Telephorus sulfureopectus*
	Black-crowned tchagra	*Tchagra senegala*
	Brown-crowned tchagra	*Tchagra australis*
	Grey-headed bush shrike	*Malaconotus blanchoti*

Family	Common Name	Scientific Name
Sturnidae	Black-bellied starling	*Lamprotornis corruscus*
Nectariniidae	Purple-banded sunbird	*Nectarinia bifasciata*
	Grey sunbird	*Nectarinia veroxii*
	Scarlet-chested sunbird	*Nectarinia senegalensis*
	Olive sunbird	*Nectarinia olivacea*
	Collared sunbird	*Anthreptes collaris*
	Amethyst sunbird	*Necarinia amethystina*
Zosteropidae	Cape white-eye	*Zosterops capensis*
Ploceidae	Thick-billed weaver	*Amblyospiza albifrons*
	Forest weaver	*Ploceus bicolor*
	Spectacled weaver	*Ploceus ocularis*
	Lesser masked weaver	*Ploceus intermedius*
	Village weaver	*Ploceus cucullatus*
	Yellow weaver	*Ploceus subaureus*
	Red-shouldered widow	*Euplectes axillaris*
Estrillidae	Green twinspot	*Mandingoa nitidula*
	African firefinch	*Lagonosticta rubricata*
	Blue waxbill	*Uraeginthus angolensis*
	Common waxbill	*Estrilda astrild*
	Grey waxbill	*Estrilda perreini*
	Bronze mannikin	*Spermestes cucullatus*
	Red-backed mannikin	*Spermestes bicolor*
Viduadae	Pin-tailed whydah	*Vidua macroura*
Fringillidae	Yellow-fronted canary	*Serinus mozambicus*
	Cape canary	*Serinus canicollis*
	Brimstone canary	*Serinus sulphuratus*

Appendix 3. The relative densities of bird species detected using the line transect and point transect methods across eight known age regenerating sites and a reference site

Relative densities	Reference site		28 year old site		24 year old site		20 year old site		16 year old site		12 year old site		8 year old site		4 year old site		1 year old site	
	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect
Black-bellied starling	0.023±0.002	0.010±0.001	0.030±0.0031	0.038±0.0099	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Black-eyed bulbul	0.043±0.002	0.049±0.003	0.095±0.0043	0.076±0.0178	0.111±0.0042	0.035±0.0029	0.058±0.0041	0.485±0.0467	0.086±0.0046	0.091±0.0233	0.093±0.0071	0.097±0.0094	0.077±0.0049	0.058±0.0036	0.064±0.0038	0.035±0.0026	0.047±0.0039	0.033±0.0020
Bleating warbler	0.100±0.002	0.099±0.003	0.345±0.0053	0.248±0.0567	0.279±0.0052	0.312±0.0069	0.277±0.0052	2.426±0.0606	0.329±0.0065	0.409±0.1014	0.267±0.0092	0.371±0.0074	0.221±0.0082	0.275±0.0086	0.179±0.0053	0.129±0.0057	0.030±0.0031	0.023±0.0017
Blue-grey flycatcher	0.002±0.004	0.010±0.001	0.030±0.0027	0.017±0.0042	0	0.018±0.0043	0	0	0	0	0	0	0	0	0	0	0	0
Blue-mantled flycatcher	0.023±0.004	0.016±0.002	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brown robin	0.005±0.006	0.005±0.000	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brown-hooded kingfisher	0.011±0.0008	0	0.006±0.001	0.013±0.0033	0.005±0.0010	0	0	0	0	0	0	0	0	0	0	0	0	0
Burchells coucal	0	0.005±0.0008	0.030±0.0022	0.017±0.0042	0.011±0.0014	0.006±0.0014	0	0	0	0	0	0	0	0.007±0.0021	0.038±0.0035	0.035±0.0033	0.006±0.0013	0.007±0.0012
Cape wagtail	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.003±0.0009
Cape white eye	0	0	0	0	0.016±0.0030	0	0	0.485±0.1041	0	0	0	0	0	0.058±0.0125	0	0.045±0.0082	0	0
Collared sunbird	0	0.010±0.0011	0.024±0.0025	0.021±0.0054	0.011±0.0014	0.024±0.0026	0.015±0.0021	0.139±0.0280	0	0	0.058±0.0065	0.032±0.0038	0	0	0	0	0	0
Common quail	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.010±0.0014
Common waxbill	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.167±0.252	0.252±0.0325	0.101±0.0162	0.370±0.0390
Diederick cuckoo	0.007±0.0006	0.003±0.0006	0.006±0.0011	0.008±0.0024	0.000±0	0.012±0.0020	0	0	0	0	0	0	0	0.022±0.0033	0.013±0.0019	0.020±0.0024	0.018±0.0021	0.020±0.0017
Dusky flycatcher	0.002±0.0004	0	0	0	0.005±0.0010	0	0.007±0.0015	0	0	0	0	0	0	0	0	0	0	0
Emerald cuckoo	0.011±0.0008	0.010±0.0011	0.024±0.0020	0.004±0.0014	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Emerald-spotted dove	0	0	0	0	0.005±0.0010	0.024±0.0026	0	0	0	0	0	0	0	0	0	0		0
European cuckoo	0.002±0.0004	0.003±0.0006	0	0	0	0	0	0	0	0	0	0	0	0	0	0		0
Fan-tailed flycatcher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.006±0.0013	0
Fiscal shrike	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Forest weaver	0.050±0.0029	0.021±0.0022	0.083±0.0048	0.038±0.0092	0.053±0.0047	0.047±0.0042	0.066±0.0064	0.554±0.0699	0.043±0.0046	0.100±0.0278	0.151±0.0129	0.008±0.0022	0.019±0.0044	0.036±0.0037	0	0	0	0.003±.0009
Fork-tailed drongo	0	0	0	0	0	0	0.007±0.0015	0	0	0	0	0	0	0	0.013±0.0019	0	0	0
Golden-rumped tinkerbird	0.096±0.0015	0.115±0.0041	0.250±0.0038	0.134±0.0308	0.079±0.0037	0.112±0.0077	0.015±0.0021	0.277±0.0350	0	0	0	0	0	0	0	0	0	0
Golden-tailed woodpecker	0.014±0.0009	0.013±0.0012	0.012±0.0015	0.017±0.0042	0.021±0.0019	0.018±0.0023	0	0	0	0	0	0	0	0	0	0	0	0
Grassveld pipit	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.036±0.0042	0
Green coucal	0.002±0.0004	0.010±0.0011	0.018±0.0018	0.008±0.0024	0	0	0	0	0	0	0	0	0	0	0	0	0	0.046±0.0023
Grey sunbird	0.009±0.0007	0.008±0.0011	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Relative densities	Reference site		28 year old site		24 year old site		20 year old site		16 year old site		12 year old site		8 year old site		4 year old site		1 year old site	
	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect
Klaas cuckoo	0.007±0.0006	0.005±0.0008	0.018±0.0018	0.008±0.0024	0.011±0.0014	0.012±0.0020	0.015±0.0021	0	0	0	0	0	0	0.022±0.0033	0	0	0	0
Little bee eater	0	0	0	0	0	0		0	0	0	0	0	0	0	0	0	0.030±0.0063	0.007±0.0012
Livingstone’s turaco	0.014±0.0009	0.023±0.0013	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Natal robin	0.027±0.0012	0.016±0.0013	0.071±0.0029	0.017±0.0042	0.016±0.0017	0.006±0.0014	0.029±0.0028	0.277±0.0350	0.021±0.0023	0.027±0.0077	0.023±0.0037	0.048±0.0041	0.077±0.0049	0.036±0.0037	0.019±0.0023	0.015±0.0022	0	0
Olive bush shrike	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Olive sunbird	0.023±0.0010	0.023±0.0016	0.024±0.0020	0.017±0.0042	0.021±0.0019	0.006±0.0014	0.015±0.0021	0.139±0.0280	0.014±0.0019	0.036±0.0099	0.023±0.0037	0.024±0.0034	0.010±0.0022	0.022±0.0033	0	0	0	0
Paradise flycatcher	0	0	0	0	0	0	0.022±0.0025	0	0.007±0.0014	0	0	0	0	0	0	0	0	0
Pin-tailed whydah	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.006±0.0014	0	0.113±0.0143	0.007±0.0012
Puffback	0.043±0.0021	0.029±0.0031	0.137±0.0048	0.042±0.0104	0.105±0.0052	0.147±0.0093	0.139±0.0074	0.277±0.0350	0.107±0.0061	0.127±0.0334	0.093±0	0.040±0.0040	0.048±0.0044	0.043±0.0049	0	0	0	0
Purple-banded sunbird	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Purple-crested turaco	0.048±0.0013	0.026±0.0013	0.024±0.0025	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pygmy kingfisher	0.002±0.0004	0	0	0	0	0	0	0	0	0	0	0	0	0.007±0.0021	0	0	0	0
Rattling cisticola	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.154±0.0059	0.149±0.0068	0.178±0.0065	0.182±0.0044
Red-chested cuckoo	0	0	0.006±0.0011	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Red-backed mannikin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.148±0.0316	0
Red-eyed dove	0.002±0.0004	0.010±0.0011	0	0.017±0.0042	0	0	0	0	0	0	0	0	0	0	0.006±0.0014	0	0.018±0.0038	0.033±0.0059
Rudd’s apalis	0	0	0	0	0	0	0	0	0.014±0.0027	0	0	0.121±0.0074	0.115±0.0086	0.051±0.0048	0.013±0.0019	0	0	0
Scaly-throated honeyguide	0.005±0.0005	0.005±0.0008	0.012±0.0015	0.008±0.0024	0	0	0	0	0	0	0.058±0.0053	0	0	0	0	0	0	0
Sombre bulbul	0.137±0.0014	0.203±0.0054	0.048±0.0030	0.059±0.0139	0.016±0.0017	0.029±0.0028	0.007±0.0015	0.208±0.0324	0.007±0.0014	0.000±0	0.000±0	0.048±0.0041	0.125±0.0056	0.130±0.0095	0.071±0.0033	0.064±0.0040	0	0
Southern boubou	0.030±0.0015	0.018±0.0013	0.071±0.0033	0.029±0.0070	0.000±0	0.012±0.0020	0	0	0	0	0.023±0.0037	0	0.019±0.0030	0	0	0.005±0.0014	0	0
Speckled mousebird	0.005±0.0005	0	0	0		0	0	0	0	0	0	0	0	0	0	0	0	0
Spotted flycatcher	0	0	0.006±0.0011	0	0.026±0.0051	0	0	0	0	0	0	0	0	0	0	0	0	0
Spur-winged goose	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.017±0.0043
Square-tailed drongo	0.037±0.0017	0.021±0.0018	0.030±0.0022	0.025±0.0063	0.026±0.0025	0.041±0.0036	0.058±0.0041	0.277±0.0350	0.029±0.0026	0.082±0.0218	0.047±0.0049	0.048±0.0041	0.010±0.0022	0.051±0.0048	0	0.010±0.0019	0	0
Stonechat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tambourine dove	0.014±0.0009	0.018±0.0013	0.018±0.0023	0.029±0.0072	0.005±0.0010	0.012±0.0020	0	0	0	0	0	0	0.010±0.0022	0	0.013±0.0019	0.010±0.0019	0	0
Tawny-flanked prinia	0.000±0	0.010±0.0014	0.012±0.0015	0.008±0.0024	0.005±0.0010	0	0	0	0	0.009±0.0033	0.035±0.0044	0.008±0.0022	0.029±0.0036	0.029±0.0047	0.090±0.0032	0.059±0.0032	0.036±0.0033	0.010±0.0014
Terrestrial bulbul	0.007±0.0006	0.036±0.0034	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Relative densities	Reference site		28 year old site		24 year old site		20 year old site		16 year old site		12 year old site		8 year old site		4 year old site		1 year old site	
	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect	Point transect	Line transect
Thick-billed weaver	0.005±0.0005	0	0	0	0	0	0	0	0.029±0.0043	0	0	0	0	0	0.013±0.0019	0.020±0.0024	0.059±0.0095	0.083±0.0095
Trumpeter hornbill	0.062±0.0030	0.042±0.0031	0.048±0.0052	0.017±0.0042	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wattle-eyed flycatcher	0.027±0.0015	0.016±0.0013	0.006±0.0011	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
White-browed robin	0	0	0	0	0	0	0	0	0	0	0.023±0.0037	0.016±0.0029	0.029±0.0036	0	0.038±0.0029	0.030±0.0026	0	0
White-eared barbet	0.037±0023	0.018±0.0013	0.107±0.0064	0.025±0.0066	0	0	0	0	0	0	0	0	0	0	0	0.005±0.0014	0	0
Woodward's batis	0.014±0010	0.010±0.0011	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Yellow-eyed canary	0.021±0021	0.005±0.0008	0.018±0.0023	0.004±0.0014	0.011±0.0014	0	0.036±0.0074	0	0	0	0	0.056±0.0076	0	0.029±0.0035	0	0	0.101±0.0110	0.043±0.0066
Yellow weaver	0.005±0.0005	0	0.006±0.0011	0	0	0	0	0	0.029±0.0043	0.045±0.0165	0.012±0.0027	0.008±0.0022	0	0.029±0.0036	0.051±0.0073	0.089±0.0096	0.178±0.0148	0.079±0.0082
Yellow-bellied bulbul	0.080±0.0019	0.034±0.0020	0.125±0.0044	0.017±0.0042	0.084±0.0033	0.082±0.0063	0.036±0.0030	0.416±0.0477	0.064±0.0034	0.036±0.0099	0.023±0.0037	0.024±0.0034	0.067±0.0048	0.043±0.0038	0.038±0.0045	0.025±0.0025	0	0
Yellow-breasted apalis	0	0.026±0.0026	0.048±0.0045	0.038±0.0093	0.074±0.0061	0.047±0.0047	0.161±0.101	0.277±0.0467	0.200±0.0106	0.036±0.0099	0.081±0.0069	0.024±.0065	0.067±0.0080	0	0.013±0.0019	0.005±0.0014	0	0
Yellow-spotted nicator	0	0.016±0.0013	0.060±0.0028	0	0.005±0.0010	0	0.036±0.0030	0.208±0.0324	0.021±0.0023	0	0.035±0.0044	0.024±0.0034	0.077±0.0049	0.051±0.0057	0	0	0	0
Yellow-throated longclaw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.006±0.0013	0.026±0.0035

Appendix 4. Presence absence data for point transect and breeding assemblages across eight regenerating sites and reference site. Point transect represents those species detected using point transects and breeding represents those species detected using nest quadrats. 1 denotes presence and 0 denotes absence.

Species	Reference site		28 year site		24 year site		20 year site		16 year site		12 year site		8 year site		1 year site	
	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding
Black-bellied starling	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Black-eyed bulbul	1	1	1	1	1	1	1	1	1	1	1	1	1	0	1	0
Bleating warbler	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	0
Blue-grey flycatcher	1	1	1	0	0	0	0	0	0	0	0	1	0	0	0	0
Blue-mantled flycatcher	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brown robin	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brown-hooded kingfisher	1	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0
Burchell’s coucal	0	0	1	0	1	0	0	0	0	0	0	0	0	0	1	1
Cape wagtail	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cape white eye	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0
Collared sunbird	0	1	1	0	1	0	1	1	0	1	1	1	0	0	0	0
Common quail	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Common waxbill	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Dusky flycatcher	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0
Emerald-spotted dove	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Fan-tailed flycatcher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Fiscal shrike	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Forest weaver	1	1	1	1	1	1	1	0	1	0	1	1	1	0	0	0
Fork-tailed drongo	0	1	0	1	0	0	1	0	0	0	0	0	0	0	0	0
Golden-rumped tinkerbird	1	1	1	1	1	1	1	1	0	0	0	0	0	0	0	0
Golden-tailed woodpecker	1	1	1	0	1	0	0	0	0	0	0	0	0	0	0	0
Grassveld pipit	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0

Species	Reference site		28 year site		24 year site		20 year site		16 year site		12 year site		8 year site		1 year site	
	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding
Green coucal	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Grey sunbird	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Little bee eater	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Livingstone’s turaco	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Natal robin	1	1	1	1	1	0	1	0	1	0	1	0	1	0	0	0
Olive bush shrike	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Olive sunbird	1	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0
Paradise flycatcher	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0
Puffback	1	0	1	0	1	0	1	0	1	0	1	0	1	0	0	0
Purple-banded sunbird	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Purple-crested turaco	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Pygmy kingfisher	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rattling cisticola	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Red-backed mannikin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Red-eyed dove	1	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Rudd’s apalis	0	0	0	0	0	0	0	0	1	0	0	1	1	1	0	0
Sombre bulbul	1	1	1	0	1	1	1	0	1	0	0	0	1	0	0	0
Southern boubou	1	0	1	0	0	0	0	0	0	0	1	0	1	0	0	0
Speckled mousebird	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spotted flycatcher	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0
Spur-winged goose	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Square-tailed drongo	1	1	1	1	1	0	1	0	1	1	1	1	1	0	0	0
Stonechat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tambourine dove	1	1	1	1	1	0	0	0	0	0	0	0	1	0	0	0

Species	Reference site		28 year site		24 year site		20 year site		16 year site		12 year site		8 year site		1 year site	
	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding	Observed	Breeding
Tawny-flanked prinia	0	0	1	0	1	1	0	0	0	0	1	0	1	0	1	0
Terrestrial bulbul	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thick-billed weaver	1	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0
Trumpeter hornbill	1	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Wattle-eyed flycatcher	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
White-browed robin	0	0	0	0	0	0	0	0	0	0	1	1	1	0	0	0
White-eared barbet	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Woodward’s batis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Yellow-eyed canary	1	0	1	1	1	1	1	0	0	0	0	0	0	0	1	0
Yellow weaver	1	0	1	0	0	0	0	0	1	0	1	0	0	0	1	0
Yellow-bellied bulbul	1	1	1	1	1	1	1	1	1	1	1	1	1	0	0	0
Yellow-breasted apalis	0	1	1	1	1	0	1	1	1	1	1	0	1	0	0	0
Yellow-spotted nicator	0	0	1	0	1	0	1	0	1	0	1	0	1	0	0	0
Yellow-throated longclaw	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0