The effect of habitat edges on the rehabilitation of coastal dune community structure

by

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Dedicated to my parents

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Introduction

Background

During the last decade large-scale tropical deforestation led to the fragmentation of forests, leaving islands of natural habitats (Feinsinger 1994), with relatively small functional core areas (Fraver 1994). The biological and physical consequences of deforestation are only starting to be understood by learning how such changes affect the function and structure of ecosystems (Bierregaard, Lovejoy, Kapos, Augusto dos Santos & Hutchings 1992). Large scale environmental threats like the use of deforested land for industrial forestry or human settlements (Finegan 1996), urged nature conservationists to change from an idealistic philosophical conservation view to a more practical and realistic ecological management view.

Through secondary succession, the recolonization of deforested land offers the hope that some of the unique characteristics of these fragmented ecosystems may be restored (Finegan 1996). Restoration is defined as the process where a disturbed site is intentionally altered to establish an indigenous ecosystem (Society for Ecological Restoration, 1991). To restore a habitat successfully, the original species richness, species composition and species diversity must be regained (Finegan 1996). The initial site conditions influence the species richness of secondary vegetation during the restoration process (Finegan 1996). Frugivorous species may, for example, use remnant trees or trees in the vicinity to facilitate the recolonization of deforested land (Finegan 1996). However, in spite of restoring species richness, the species compositions of restored areas may differ markedly from that of mature forests on comparable sites for many decades (Finegan 1996). This may be due to the influence of edge effects on the restoring ecosystem (Turner & Corlett 1996; Turner 1996).

In ecological terms an edge is defined as a zone where two plant communities meet, or where successional stages within plant communities come together (Noss 1983). Abutting communities may influence each other, especially at the boundary. Hence an edge effect may be seen as a change in measurable community macro - variables (e. g. species richness and species diversity) with increasing distance from the edge.

According to Leopold (1933), the greater the number of habitat boundaries (edges) present in an area, the greater the diversity and richness of species. Numerous edge effect studies have been conducted during the last 20 years on both biotic and abiotic variables (see Lovejoy, Bierregaard, Rylands, Malcolm, Quintela, Harper, Brown, Powell, Powell, Schubart & Hays 1986; Bierregaard *et al.* 1992; Bolger, Alberts, Sauvajot, Potenza, McCalvin, Tran, Mazzoni & Soulé 1997). In spite of all of these studies, little consensus has been reached on how to measure an edge and on how deleterious the edge effect can be for biological and abiotic variables (Murcia 1995). For some species the edge may be advantageous while the same edge may be detrimental to others (Howe 1984; Turner 1996). Accordingly, three categories of species responses to edges have been identified: increasing species density near the edge, decreasing species density near the edge and species which show little change near the edge (Kroodsma 1982, Sisk & Margules 1993). Variations on these responses also arise depending on whether the species is a habitat generalist, occurring on both abutting edges, or a habitat specialist, occurring commonly in only one of the two adjoining habitats (Sisk & Margules 1993). Edge species may find habitats at the edge that are not present in either community alone (Gates & Gysel 1978).

The edge width is defined as the distance to the point where the apparent influence of the road (edge) on the community variable (e.g. species richness or species diversity) changed. Edge widths for biotic and abiotic characteristics vary considerably. A review by Murcia (1995) highlighted these variable trends, with edge effects ranging from 50 metres for abiotic variables to 600 metres for plant species composition. No edge effect studies have been conducted within a sequential successional development framework.

An edge affecting environmental variables such as wind and light penetration, may change evapotranspiration balances, understory plant water relations, photosynthetically active radiation (PAR), relative humidity and temperature (Bierregaard *et al.* 1992). Soil variables such as pH, organic carbon, total nitrogen, available phosphorus and soil moisture are also known to increase towards a forest's interior (Jose, Gillespie, George & Kumar 1996), thus resulting in a relatively fertile forest core compared to the forest edge. Increased light penetration at edges may change vegetation composition (Fox, Taylor, Fox & Williams 1997). Here an increase in weed and pioneer species at the edge (Oliveira-Filho, de Mello & Scolforo 1997) may result in an increase in species richness, while actual forest species richness may decrease (Saunders, Hobbs & Margules 1991). Although the edge may consist of many alien vegetation species (Brothers & Spingarn 1992), edge vegetation can be considered as a buffer that protects forest vegetation from conditions in adjacent clearings (Williams - Linera 1990).

There is strong evidence that more predators occur in the edge zone (Gates & Gysel 1978; Marini, Robinson & Heske 1995), which cause increases in nest predation with decreasing distance from the edge (Gates & Gysel 1978; Yahner & Wright 1985; Ratti & Reese 1988; Paton 1994; Marini, Robinson & Heske 1995).

Edge-driven changes in the forest environment may indirectly affect the inhabitants of the forest (Murcia 1995). Insects often occur more frequently near edges than further into the forest (Courtney & Courtney 1982; Cappucino & Martin 1997). This may be due to the increased amount of sunlight at the edge compared to the forest interior, causing an increase in the nutritional value of leaves at the edge (Collinga & Louda 1989). The increase of insects at edges will lead to an increase in insectivorous birds at edges (Strelke & Dickson 1980). Furthermore, red-backed voles (*Clethrionomys calirornicus*) were found to occur at a higher density in the forest interior than at the edge, due to the higher densities of their primary food item, sporocarps of mycorrhizal fungi in the forest interior (Mills 1994).

A well developed edge vegetation community protects the forest interior from much of the variable microclimate edge effects (Chen, Franklin & Spies 1995; Kellman 1996). Humidity tends to be lower, and soil and air temperatures higher at the edges than in the centre of a forest (Rannay, Bruner & Levenson1981; Williams-Linera 1990). There is also a decrease in solar radiation, light penetration and wind exposure from the edges to the interior of the forest (Rannay *et al.* 1981). Thus, the forest interior has a more stable microclimate than areas near to the forest edge due to the influence of edge effects.

After many years of studies on various taxa and species, there are still no clear guidelines to predict edge effects. This lack of consistency may be due to the physiognomy (Matlack 1993; Matlack 1994), orientation (Brothers & Spingarn 1990; Fraver 1994; Matlack 1994), level of disturbance (Fox *et al.* 1997), habitat type surrounding edges and the age of edges (Saunders *et al.* 1991; Matlack 1994).

Physical and biological changes that occur along newly induced edges may affect the form and function of the fragmented ecosystem (Bierregaard *et al.* 1992).These newly made edges will affect the landscape which, in turn, can have important influences on the biota, especially those occurring in the area nearest to the edge and the surrounding agricultural or developed land (Saunders *et al.* 1991). A habitat fragment is likely to have more species than it may be able to maintain and species will then be lost. This process of "species relaxation" is considered an inevitable consequence of fragmentation (Saunders *et al.* 1991). However, the loss of species will depend on the physical changes caused, and their effect on the biotic interactions.

Constantly changing forest edges have caused large numbers of birds being forced to find a new habitat in the forest interior (Hagan, Van der Haegen & McKinley 1996). This increase in species richness and density within a fragment shortly after creating an edge, has been described as "new species packing into remaining habitats" (Hagan *et al.* 1996). In contrast to the results of the previous study on newly made edges, the species densities and richness at older edges did not differ as much from the forest interior as those in the more recently created edges (Matlack 1994). It must however, be noted that species density can increase in fragmented systems due to alien species and edge specialists, but concurrently the density of species originally found in the area may continue to decline (Saunders *et al.* 1991). Areas fragmented for long periods will lose some of their original species, but could gain additional species that would not have colonised in the area before fragmentation (Hagan *et al.* 1996).

Most edge-related studies have concentrated on edges between an agricultural land and a forest, with only a few studies assessing the influence of roads as a barrier (Oxley, Fenton & Carmody 1974; Adams & Geis 1983; Mader, Schnell & Kornacker 1990). A narrow, seldom - used vehicle path of three metres can have a significant effect on small mammals (Swihart & Slade 1984) and results from Oxley *et al.* (1974) suggest that road surface may not be a critical inhibiting factor for the movements of small mammals.

Rationale for present study

Some of the second and third dunes from the coastline between Richards Bay (28°43′S, 32°12′E) and the Mapelane Nature Reserve (32°25′S, 28°27′E) are presently being mined by Richards Bay Minerals (RBM). After mining, a third of the area is rehabilitated to an indigenous forest, while the remaining two-thirds of the area is revegetated with *Casuarina equisetifolia* (beefwood) for the development of a local charcoal industry (Van Aarde, Ferreira, Kritzinger, Van Dyk, Vogt & Wassenaar 1996a). Rehabilitation started 20 years prior to the present study and has resulted in the development of a known age sere of coastal dune forest succession (Van Aarde *et al.* 1996a). The different aged sites (seral stages) vary in size and shape and are bordered either by unmined indigenous forest, beefwood plantations, bare mined sand dunes or other regenerating dune forests. A network of roads and tracks run through the rehabilitating sites, and divides them into smaller fragments. Such fragmentation may affect rehabilitation success as discussed earlier. The present study, therefore, aims at estimating the implications of the roads passing through the rehabilitating forests by considering them as edges. Because of the significance of the edge - effect concept to wildlife management in the planning of reserves, this study was initiated to evaluate responses of bird, millipede and rodent communities to roads as edges in relation to successional development.

These taxa were selected because studies on their development during dune rehabilitation have recently been undertaken in the same study sites as the present study (Ferreira 1993; Kritzinger 1996; Van Aarde, Ferreira & Kritzinger 1996b). They furthermore represent different levels in the developing coastal dune food web and have different functions in the ecosystems.

Objectives of study

To determine whether bird, millipede and rodent communities are affected by roads as edges, and whether these edges change with habitat regenerating age.

To achieve these objectives, the following hypotheses were raised:

$H_0$1: Distance from edges (roads) does not affect species richness, diversity and number of individuals of rehabilitating coastal dune forest bird, millipede and rodent communities.

$H_0$2: Habitat regeneration age does not influence edge widths for bird, millipede and rodent communities.

Key question for Hypothesis 1:

1) Do the species richness, diversity and number of individuals of bird, millipede and rodent communities change from the edge to the interior of the rehabilitating forest fragments?

Key question for Hypothesis 2:

1) Are edge widths for bird, millipede and rodent richness, diversity and number of individuals affected by the regeneration age of habitats?

Chapter 2

Materials and methods

Introduction

The present study deals with several species of three taxa. Thus it is essential to investigate every taxon on its own but to analyse the data on the same level. This chapter describes the study area, general data collection and the general statistical methods used to analyse data for bird, millipede and rodent assemblages. Aspects unique to each of these assemblages will be presented in the relevant chapters.

Study area

The study area is situated on the southern end of the Mozambique coastal plain, which is 8.6 km north and 9.3 km east of Richards Bay (28°43′S, 32°12′E), and stretches to the Mapelane Nature Reserve (32°25′S, 28°27′E) (Figure 1). The rehabilitating areas where the study took place are on the seaward side bordered by a relatively narrow (100 - 250 metres wide) belt of unmined fragmented patches, and on the interior, by a strip of beefwood plantations which do not invade the regenerating areas. These areas have a long history of human presence and disturbances (Van Aarde *et al.* 1996a).

The area has a humid climate with the mean annual temperature at Cape St. Lucia Station being 21. 5 °C (Weisser 1987) as well as a high mean annual rainfall of 1292 mm (Van Aarde *et al.* 1996a). The vegetation of unmined coastal dunes comprises 50 to 60% thicket species and the rest are forest species (Tinley 1985).The nature of these forests depends on their distances from the high-tide mark and whether or not they face sea winds (Pooley 1994).

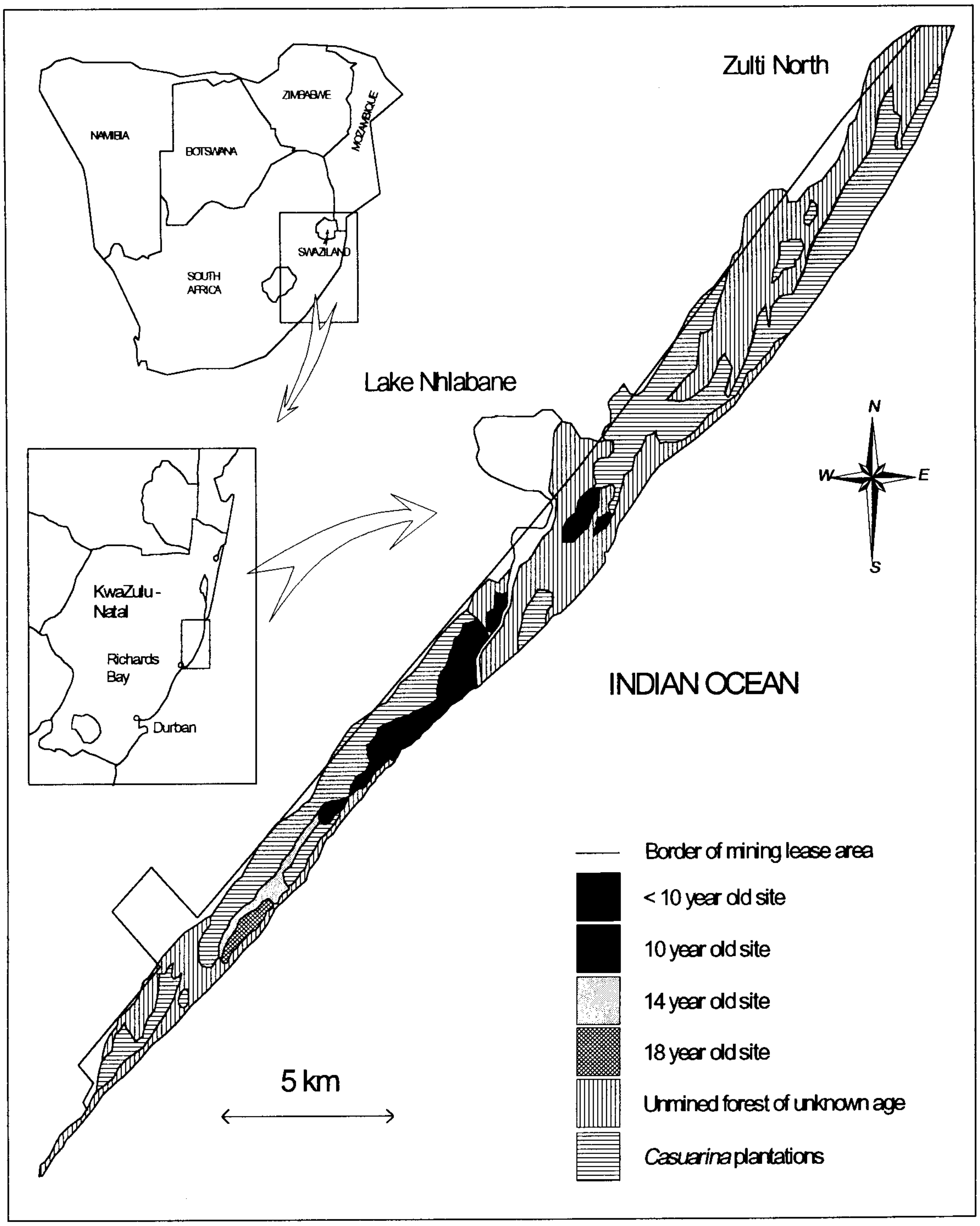

Figure 1 Map of the study area north of Richards Bay (28°43'S, 32°12'E), illustrating location, study sites, unmined forest and *Casuarina* plantations as they appeared when fieldwork was conducted.

Since July 1977, Richards Bay Minerals have undertaken open cast mining of the second and third dunes north of Richards Bay for heavy metals such as zircon, ilmenite and rutile (Van Aarde *et al.* 1996a). The mining is preceded by the complete destruction of dune vegetation. Rehabilitation was initiated here between 1978 and 1980. The rehabilitation process has been described by Van Aarde *et al.* (1996a) and several other authors (Weisser 1987; Ferreira 1993; Kritzinger 1996; Van Dyk 1996; Ferreira 1997).

This rehabilitation process commenced 20 years prior to the present study, and has given rise to a series of known aged sites. Four of these and an unmined area of unknown age have been selected as study sites. These sites are bordered either by unmined indigenous forest, bare mined sand dunes, other regenerating dune forests and, on the hinterland, by beefwood plantations. The unmined forest of unknown age is situated about 30 km north of the ecology centre of RBM which is located near the southern end of the rehabilitating area (Van Aarde *et al.* 1996a). A network of roads and tracks run through these rehabilitating sites and divides them into smaller fragments (Figure 2). Vegetation variables for each of these sites are presented in Table 1.

In the study area age related regeneration of coastal sand dune vegetation from open grasslands, to shrub lands, to a forest over a period of 20 years is associated with the development of a bird community from grassland generalists to forest specialists (Kritzinger & Van Aarde 1998). Here richness and diversity increase with an increase in structural heterogeneity (Kritzinger & Van Aarde 1998) with 57 species recorded during the summer months (L.J. Niemand, Department of Zoology and Entomology, University of Pretoria, South Africa *in litt.*). Amongst millipedes diversity increased from two to ten species with regeneration age (Van Aarde *et al.* 1996b), while a more recent study showed that the 20 year old community comprised 16 morphospecies (M. J. Neluvhalani, Department of Zoology and Entomology, University of Pretoria, South Africa *in litt.*). The post - mining development of the rodent community is characterised by an overall decline in density and an increase in diversity with regeneration age. Initial domination of grasslands by the multi - mammate mouse (*Mastomys natalensis*) as a pioneer species is followed by woodlands dominated by the pouched mouse (*Saccostomus campestris*) and the red veld rat (*Aethomys chrysophilus*) (Ferreira & Van Aarde 1997). Rodents other than these occur at very low densities in the study area (Ferreira 1993).

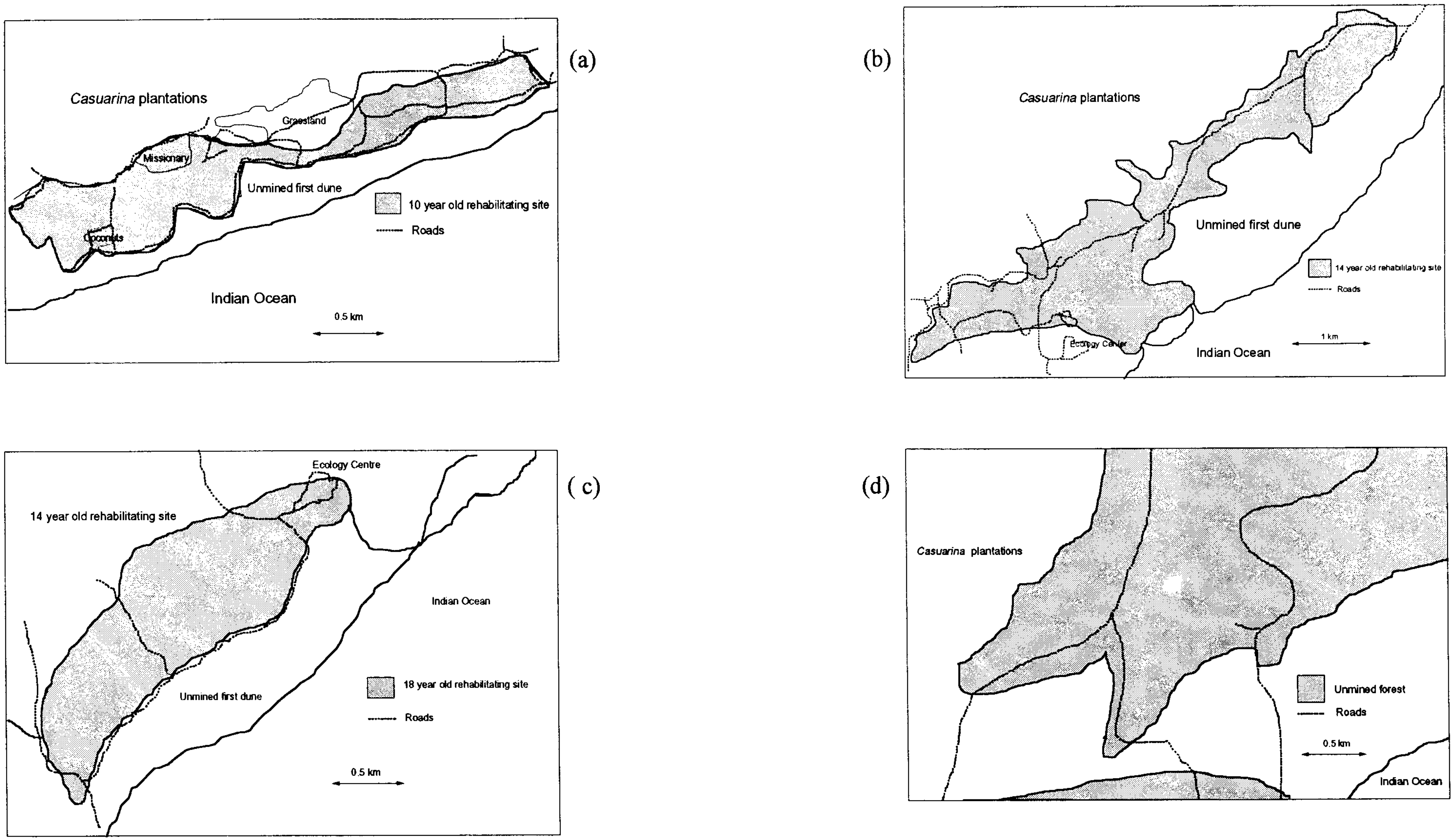

Figure 2 Line diagrams of road networks on each of the study sites where (a) represents the 10 year old rehabilitating site, (b) the 14 year old rehabilitating site, (c) the 18 year old rehabilitating site and (d) the unmined forest.

Table 1 A description of the vegetation in the rehabilitating and unmined sites in the study area (adapted from Van Aarde *et al.* 1996a).

Age of the site in years	Description
10	Rehabilitating area comprising an *Acacia karroo* (sweet thorn) woodland 8 - 10 m high. The canopy is dominated by *Acacia karroo* while other woody species such as *Brachylaena discolor* (coast silver oak) and *Vepris lanceolata* (white ironwood) is also present. The ground cover is dominated by *Digitaria diversinervis* (finger grass).
14	*Acacia karroo* woodland dominate with a canopy of 9 - 12 m high and secondary dune forest tree species colonizing. These include *Trichilia emetica* (Natal Mahogany), *Celtis africana* (white stinkwood), *Vepris lanceolata* and *Kraussia floribunda* (rhino - coffee). Ground cover is dominated by *Digitaria diversinervis.*
18	Rehabilitating area comprising an *Acacia karroo* woodland with 10 - 14 m high canopy. Other canopy trees include *Mimusops caffra* (coastal red milkwood), *Celtis africana, Albizia adiantifolia* (flat - crown) and *Trema orientalis* (pigeon wood). Creepers present are *Sarcostemma viminale* (caustic creeper) and *Adenia gummifera* (umPhindamshaya) with *Asystasia gangetica* (isiaobo) as one of the common ground cover species.
Unknown (unmined forest)	Unmined area comprising a secondary coastal dune forest with a canopy of 12 - 15 metres or higher. Canopy species include *Celtis africana, Trichilia emetica* and *Albizia adiantifolia.* The middle stratum consists of species such as *Ochna natalitia* (Mickey Mouse Bush), *Tricalysia sonderiana* (coast coffee) and *Carissa bispinosa* (forest num - num). *Issoglossa woodii* (herb) occurs frequently in the shrublayer.

Two years prior to the present study, MacCullum (*in prep.*) studied the influence of edges on temperature and humidity in the rehabilitating sites. Significant edge widths for humidity and temperature were shown in the 10 and 14 year old (age of sites when fieldwork for the present study was conducted) rehabilitating coastal dune forests (Figures 3 & 4). In the 18 year old regenerating site no significant edges were identified for humidity or temperature (Figures 3 & 4). The existence of significant edge widths were explained by the removal of side vegetation to allow for the passage of vehicles, which would prevent a natural sealing of the outermost edge. Because of increased light penetration near edges, a denser band of vegetation may form behind the edge, causing an increase in humidity and temperature. In the 18 year old regenerating forest the absence of edges were ascribed to increased structural heterogeneity (MacCullum *in prep.).*

Collection of data

Description of transect lines

In each study site five clusters of sampling lines, each consisting of five transect lines, were placed perpendicular to the road (Figure 5). These lines stretched from the road to the centre of the site except for the unmined forest. These transect lines were used for bird and rodent sampling. Transect lines in clusters were 10 metres apart and the clusters within a site were at least 50 m apart. The lengths of these lines varied between 80 m and 210 m due to the variation in width of the sites. Transect lines in each cluster are not independent from one another and each cluster was thus considered as a single sample measurement (see Hansen, Howitz & Madell 1993; Greenwood 1996).

For millipede sampling, quadrats (3 X 15 metres) were layed out at the edge (0 metres), 20 m, 40 m, 60 m, 80 m, 100 m and 120 m along transect lines set out randomly throughout the site and perpendicular to the edge of the road. The sampling of transect lines continued until the coefficient of variation reached a plateau. Values levelled off at 13, 12, 19 and 14 transect lines set out in the 10, 14 and 18 year old regenerating and unmined site respectively.

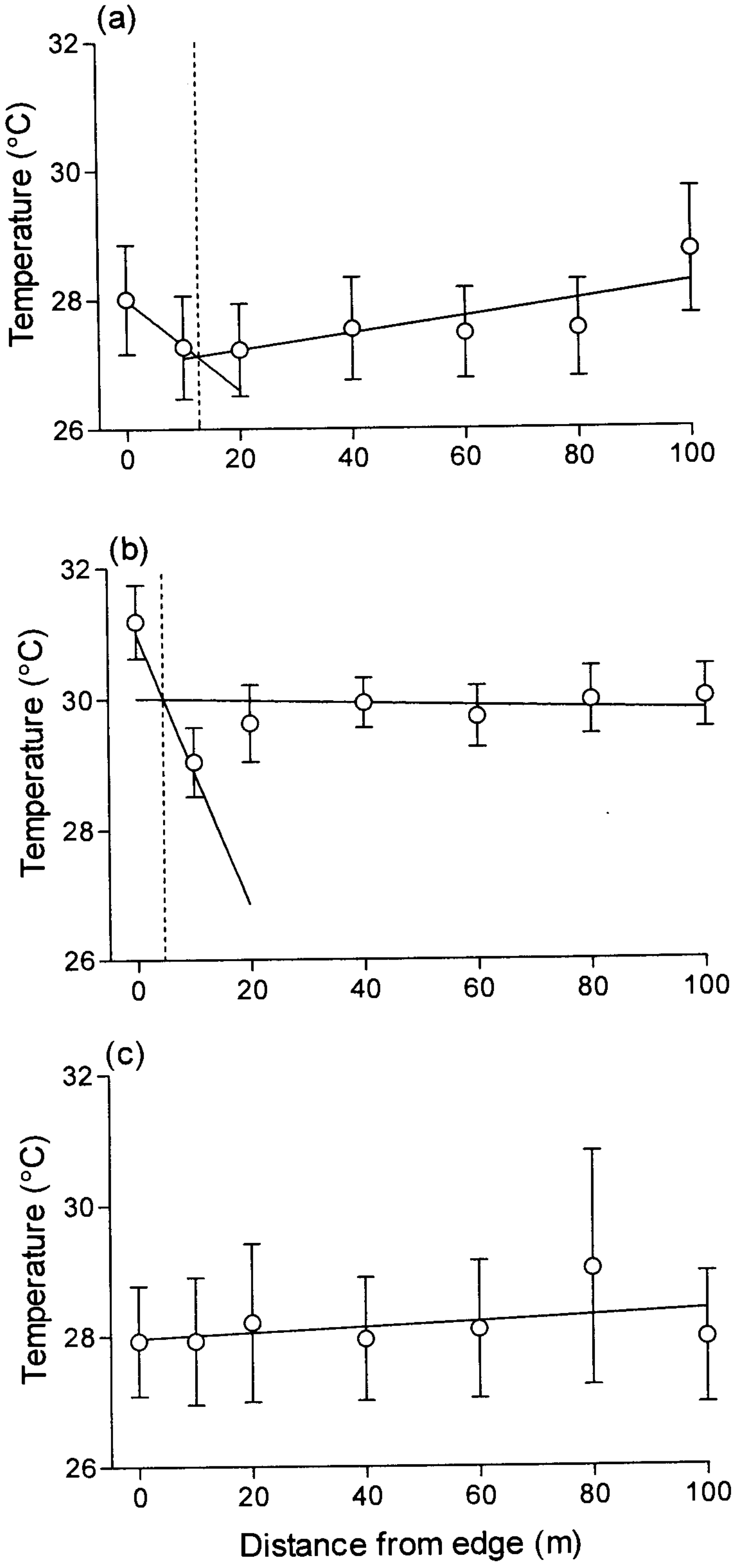

Figure 3 Mean (± standard error) temperatures in the regenerating coastal dunes forests (based on data collected by MacCullum (*in prep.*): 10 year old (a), 14 year old (b) and 18 year old site (c). The continuous vertical lines represent the standard error of mean values and the dashed vertical lines the calculated edge width. The regression lines were fitted and edge widths calculated as described on p 16.

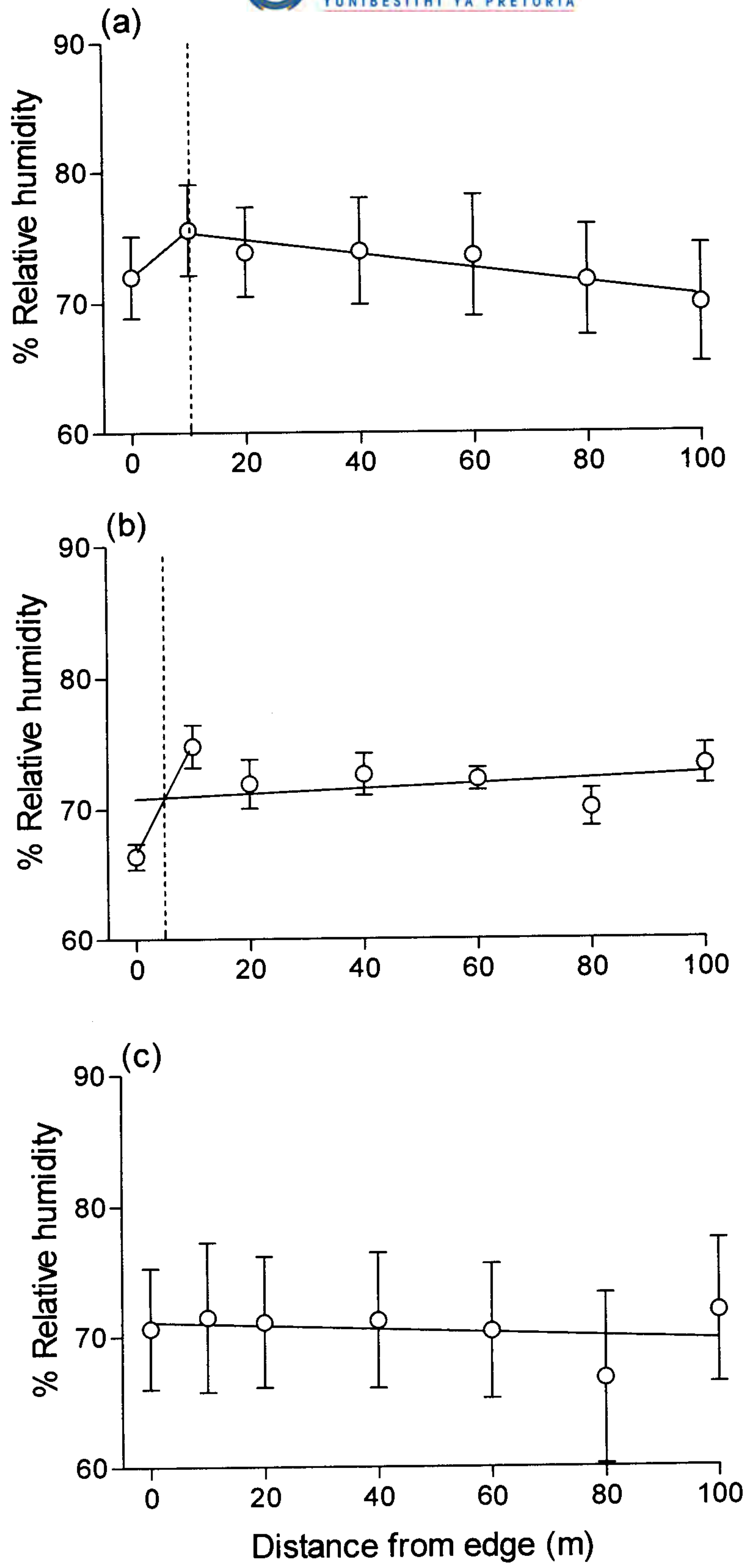

Figure 4 Mean (± standard error) % relative humidity in the regenerating coastal dunes forests (based on data collected by MacCullum (*in prep*.): 10 year old (a), 14 year old (b) and 18 year old site (c). The continuous vertical lines represent the standard error of mean values and the dashed vertical lines the calculated edge width. The regression lines were fitted and edge widths calculated as described on p 16.

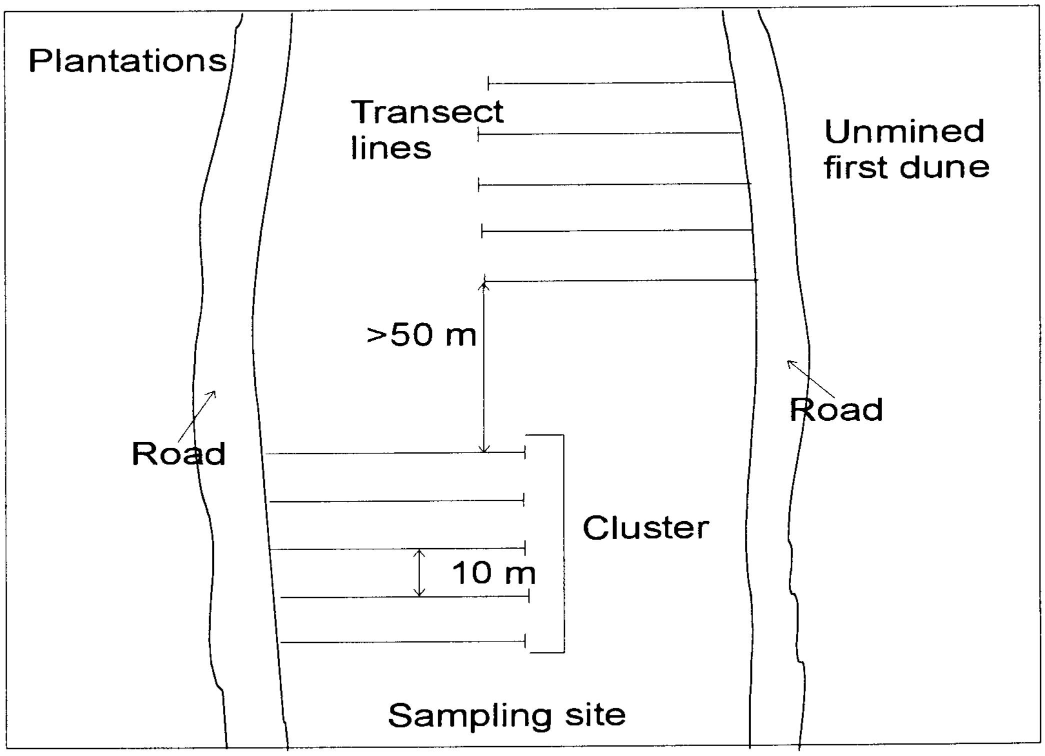

Figure 5 A line diagram to illustrate the layout of transect lines and clusters in the rehabilitating sites. The transect lines in the rehabilitating sites continued into the centre of the forest.

Data analyses

Primary analysis of the data

Data were set out in matrices, grouping the abundances of species sampled at each distance from the edge. Due to the differing lengths of the transect lines, the number of samples for each distance class varied. These data matrices were used for further calculations in the analyses.

Community variables

Species richness (S) was calculated as the mean number of species observed at every sampling distance. Shannon Wiener's diversity index ($H' = -\sum_{I} p_i (\log p_i)$), where p_i is the proportional contribution of the i^{th} species to the total count at a given sampling distance, was used to express species diversity for every sampling distance. All univariate data were tested for Gaussian distribution by obtaining a p-value from Dallal and Wilkinson's approximation of Lilliefors' method (Sokal & Rohlf 1995).

Quantifying the edge

The edge width was defined as the distance to the point where the edge (road) had an influence on a given variable. A p - value calculated by using the continuous two-phase regression model of Nickerson, Facey & Grossman (1989) as used by Matlack (1993) and Fox *et al.* (1997) indicated if a significant edge width existed or not. In this analysis, all data were divided into two groups, with all possible pairs e.g. 0 m -10 m and 20 m -120 m or 0 m - 20 m and 30 m -120 m, and error sums of squares calculated for all these possible pairs of linear regressions, with no overlap between pairs. Separate linear regressions were calculated for each of these data sets. For every regression line a value on the abscissa of the division point was estimated. The division point with (i) the lowest collective error sum of squares and (ii) a significant regression for at least one group was accepted as the best estimate of the edge width (see Matlack 1993; Fox *et al.* 1997).

The community in space

To visualise changes in community composition in different aged sites with distance from the edge, data was ordinated by non - metric multi - dimensional scaling (MDS) by using Bray - Curtis distances (Clarke & Warwick 1994). The software program PRIMER (Clarke & Warwick

1994) which was used for community analyses in space, made use of Bray - Curtis distances. It has been proved that the Bray - Curtis similarity index is robust in ecological work (Clarke 1993). Fourth - root transformation was used due to the effect in down - weighing the abundant species, so that the less dominant and rare species also play a significant role in the analyses. This transformation ensures the most biologically sensible interpretation of the multi - dimensional scaling and cluster analyses results (Clarke & Warwick 1994). The reduced dimensional ordinations have a stress value, which is a measure of how well the dimensional plot presents the actual rank similarities between the distances from the edge sampled (Clarke & Warwick 1994). Cluster analysis was used to show natural groupings between sampling distances, thus seeing if natural edge and forest core assemblages exist. Analysis of similarity was used to test for significant differences between these assemblages (ANOSIM, PRIMER, Clarke & Warwick 1994), and whether replicates from different sites vary more than replicates from within sites.

Species analyses

If edge and core assemblages occurred, the following was calculated: mean percentage contribution to the edge and core assemblage of each species, consistency and cumulative contribution of species to between edge and core dissimilarity and similarity (SIMPER, PRIMER, Clarke & Warwick 1994). This analyses method (SIMPER) consists of computing the average dissimilarity between all pairs of edge and core assemblages by comparing every species in the edge assemblage with every species in the core assemblage. This average dissimilarity is then broken down into the separate contributions from each species and the average contribution from each species to the overall dissimilarity between edge and core assemblages are calculated. Due to many pairs of species making up the average dissimilarity, it is possible to calculate the consistency of a species (Clarke & Warwick 1994). Species contributing most to differences between edge and core were identified as discriminant species and species contributing the most to within edge or core as typical edge or core species.

Chapter 3

Edge effects in the bird communities

Introduction

The influence of habitat loss and fragmentation on bird communities is of global interest (Newmark 1991) and changes in bird communities after fragmentation have been studied extensively, especially in temperate (Burgess & Sharpe 1981; Krüger & Lawes 1997) and tropical regions (Lovejoy *et al.* 1986; Newmark 1991). Habitat loss due to fragmentation may result in species richness for forest species decreasing (Saunders *et al.* 1991). Simultaneously, an increase in the overall species richness resulting from different species occupying new habitat types, cause fewer forest core bird species to occupy the habitat after fragmentation (Wiens 1989; Saunders *et al.* 1991). Thus an increase in edges may result in the loss of forest core specialist species (Yahner 1988).

The concept of edge effects is especially applicable to birds due to their higher occurrence in mixed rather than uniform habitats (Johnston 1947; Gates & Gysel 1978). Since edges are considered to be mixed habitat areas, birds breeding near edges are typical mixed - habitat breeding species (Gates & Gysel 1978). These species need open canopies with elevated observation or singing perches and a well developed understory for nesting and feeding (Gates & Gysel 1978). These mixed - habitat breeding species would then account for an increase in species nesting near edges. With newly created edges causing greater vegetative complexity, edges can have inhibiting or stimulating effects on the distribution of birds.

Some studies showed increased diversities near edges compared to areas within the forest interior, probably as a result of the increase in vegetation layers and occurrence of more than one habitat type at the edge (Strelke & Dickson 1980; Howe 1984). These vegetation differences between the edge and interior may lead to changes in microclimate (Restrepo & Gómez 1998).

After many years of studies on the influence of edge effects on birds, there are still no clear-cut guidelines to predict edge effects. It is likely that edge effects interact with each other (Murcia 1995) and the extent of edge effect depends on the species, taxa and various environmental factors (Brothers & Spingarn 1992; Fraver 1994; Matlack 1994).

Few studies have examined the influence of edge age in combination with the influence of edge effects. In Pennsylvania age of the edges had no influence on determining the rate of artificial ground nest depredations (Yahner, Morrell & Rachael 1989), but Yahner & Wright (1985) found fewer incidences of nest predation in younger than in older clear cut sites, partly due to greater vegetative cover at ground level in younger sites. In the rehabilitating coastal dune forests of KwaZulu - Natal, the bird species change with increasing regeneration age from typical grassland species, through shrubland species to typical forest species (Kritzinger & Van Aarde 1998). Large trees and dense undergrowth in the unmined forest, appeared to be responsible for differences between the bird communities of rehabilitating sites and the unmined forest (Kritzinger & Van Aarde 1998). These changes of the bird community with an increase in regenerating age support theoretical successional models (Kritzinger & Van Aarde 1998).

In this chapter birds on the individual, community and guild levels were examined. Here the effect of roads as edges and the changes of these with regeneration age will be addressed.

Materials and methods

Data collection

Transects were surveyed in random order from 22 August to 24 October 1997. Sampling started at dawn and ended at 11h00, with no observations on rainy and windy days. Sampling consisted of 12 minutes spent at a fixed location at 10 metre intervals along each of the transect lines. No observations were recorded during the first two minutes at a sampling point, thus allowing the birds to become accustomed to my presence. To standardize the sampling area, no individuals flying over the observation point were recorded. To establish the distance of the bird from the observer, an imaginary line was drawn from the bird to the ground. This distance from the observer to the spot on the ground was calculated using a "Ranging" range finder (Telinject Ranging TLR 75) calibrated at 0.5 metre intervals from 9 to 13 metres, and at one metre intervals from 13 to 40 metres. Each individual was placed in one of the distance classes (e. g. 0 m - 9 m, or 10 m - 19 m) from the edge. I assumed that all birds present were seen, that birds were recorded at their initial distance from the edge, and that all distance estimates were accurate.

Data analyses

The number of individuals of each species within a distance class on each of the transect clusters, provide a value of abundance for that cluster. Mean and associated measures of variance were calculated for each site (Hansen, Howitz & Madell 1993; Greenwood 1996). The data describing the community were analysed, edge width quantified and community analysis done as described in Chapter 2 (p 16). Percentage contribution and consistency of contribution of different bird species to between edge and core assemblages' dissimilarity and similarity were calculated using the method SIMPER as described in Chapter 2 (p 17). Species richness, diversity, number of individuals and species composition were calculated for the edge and core assemblages.

Feeding guilds

Birds were grouped into one of three feeding guilds, as either frugivores, nectarivores or insectivores based on literature (Maclean 1995) and personal observations. Frugivores are species that consumed fruit and/or seed, insectivores consumed primarily insects, and nectarivores included flower - piercing and nectar feeding species. The edge widths of all three feeding guilds in every aged site were identified using the technique of Nickerson *et al.* (1989) as described in Chapter 2 (p 16).

Results

Community variables

Based on the slopes of least square linear regression lines used to estimate edge widths, bird species richness and diversity decreased significantly ($p < 0.05$) from the edge to the interior in the regenerating sites but not in the unmined forest (Figures 6 & 7). The number of individuals decreased significantly ($p < 0.05$)from the edge to the forest interior in the 10 and 18 year old rehabilitating sites (Figure 8). The decrease in number of individuals from the edge with increase

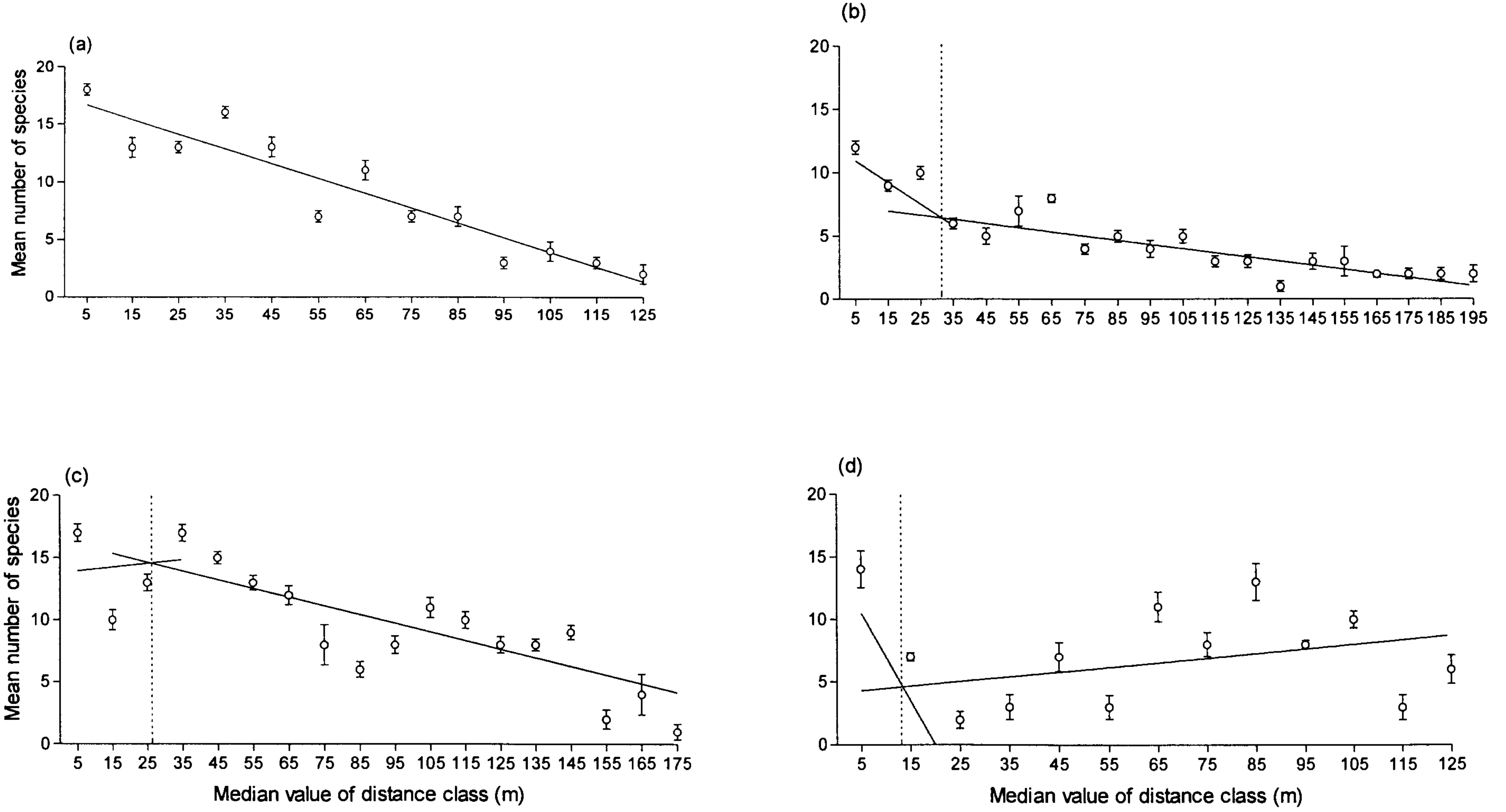

Figure 6 Mean (± standard error) species richness for birds in regenerating and unmined coastal dune forests: 10 year old site (a), 14 year old site (b), 18 year old site (c) and the unmined forest (d). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 2). The regression equations are listed in Appendix 1.

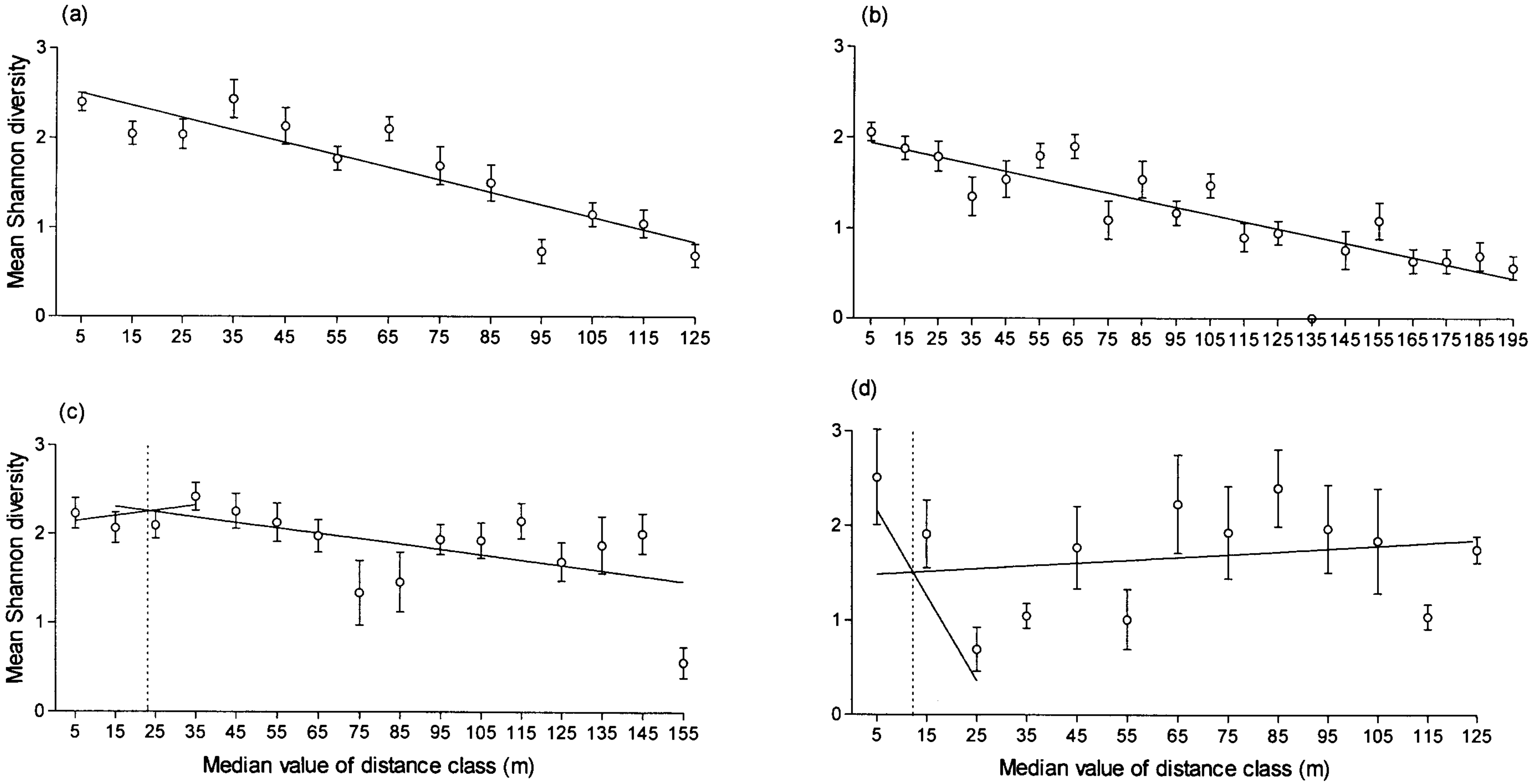

Figure 7 Mean (± standard error) Shannon diversity for birds in regenerating and unmined coastal dune forests: 10 year old site (a), 14 year old site (b), 18 year old site (c) and the unmined forest (d). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 2). The regression equations are listed in Appendix 1.

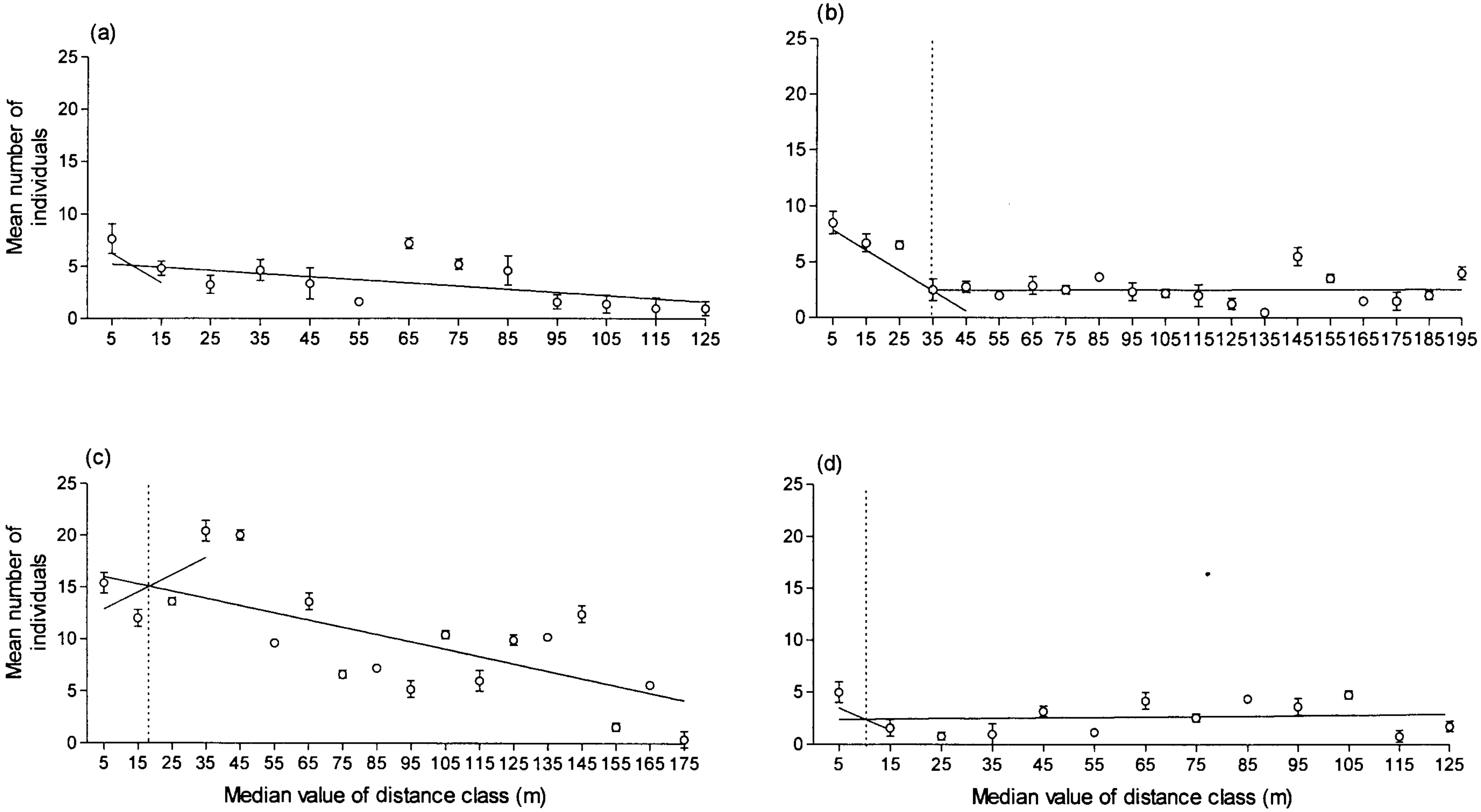

Figure 8 Mean (± standard error) number of individuals for birds in regenerating and unmined coastal dune forests: 10 year old site (a), 14 year old site (b), 18 year old site (c) and the unmined forest (d). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 2). The regression equations are listed in Appendix 1.

in distance in the 14 year old regenerating site and unmined forest did not differ significantly from zero (Figure 8). The regression equations for the community variables of all aged sites are listed in Appendix 1.

Quantifying the edge

The technique adapted from Nickerson *et al.* (1989) resulted in significant edge effects calculated for species richness and number of individuals for all the study sites except the 10 year old regenerating site. Significant edge widths for species diversity were estimated in the 18 year old rehabilitating site and unmined forest (Table 2). The edge widths for species richness, diversity and number of individuals decreased with increasing regeneration age (Figure 9).

The community in space

Based on the MDS an edge and core assemblage was identified in the 10 year old site. Here the edge assemblage extended to 75 m (median - values of distance classes) and the core assemblage from 85 m to 125 m (stress = 0.08) (Figure 10). Cluster analysis of data for the 14 year old site also distinguished between an edge and core assemblage, with the edge extending to 55 m, and the core assemblage ranging from 65 m to 195 m, stress = 0.12 (Figure 11). On the 18 year old site the edge assemblage extended from the edge to 115 m and the core assemblage ranged from 125 m to 175 m (stress = 0.15) (Figure 12). In the unmined forest the edge assemblage extended from 5 m to 25 m and the core assemblage extended from 35 m to 125 m (stress = 0.13) (Figure 13).

R - values, based on the analysis of similarity, confirmed that similarities within edge and core assemblages are smaller than between these assemblages (10 year old site, R = 0.97, $p < 0.0001$, 14 year old site, R = 0.933, $p < 0.0001$, 18 year old site, R = 0.987, $p = 0.001$, and unmined forest, R = 0.601, $p = 0.007$).

Species analyses

The mean percentage contribution of each species, either to the edge or the core assemblages (as identified by cluster analyses and multi - dimensional scaling), as well as the consistency of each species to contribute to the dissimilarity between the edge and core assemblages is presented in Table 3. Only species contributing to 50% of the cumulative total are listed. The relative

Table 2 Mean (± standard error) edge widths in metres (see text p 16) for species richness, species diversity and number of individuals for birds in each study site, followed by the *F* and *p* - values.

Community variable	Study site											
	10 year old rehabilitating site			14 year old rehabilitating site			18 year old rehabilitating site			Unmined forest of unknown age		
	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*
Species richness	-	0.9	ns	32 ± 1.5	3.2	<0.1	27 ± 1.9	5.7	<0.02	13 ± 2.2	2.1	<0.2
Species diversity	-	0.8	ns	-	1.0	ns	22 ± 1.5	19.9	<0.0001	12 ± 1.6	2.0	<0.1
Number of individuals	-	0.3	ns	35 ± 1.5	28.4	<0.0001	18 ± 1.5	6.8	<0.02	10 ± 1.6	3.3	<0.1

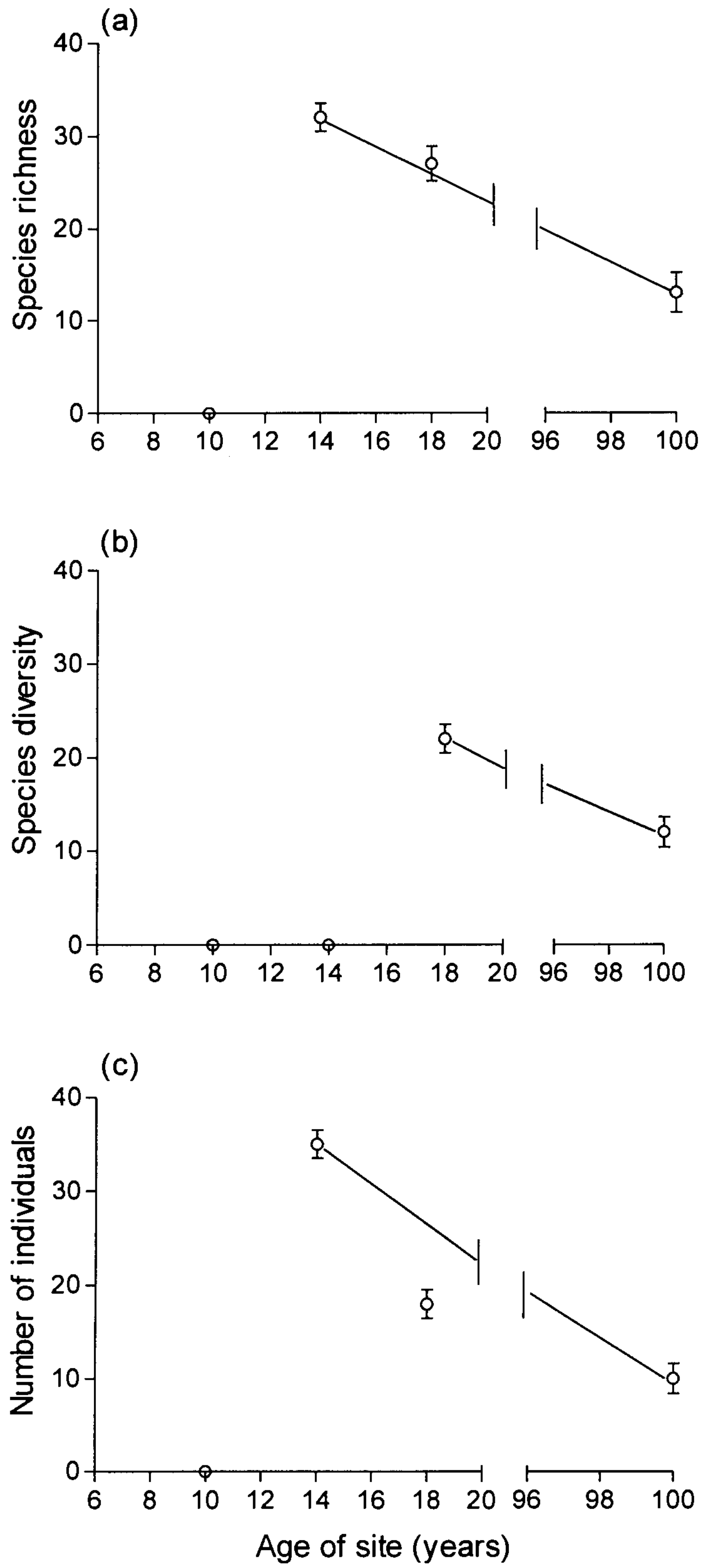

Figure 9 Mean (± standard error) edge widths for bird community variables (as described on p 16 and presented in Table 2) where (a) represents species richness, (b) species diversity and (c) the number of individuals. In this figure, the unmined forest is presented as 100 years old.

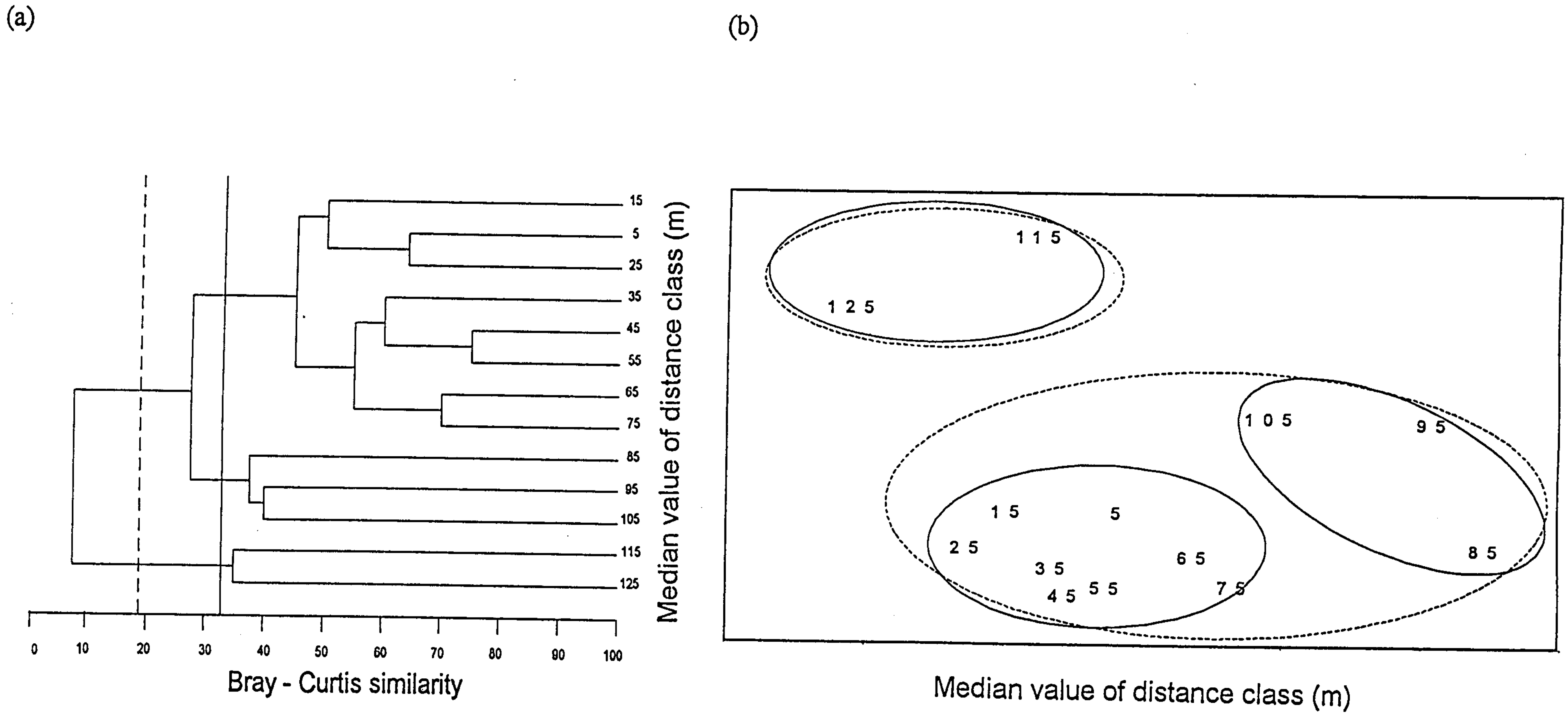

Figure 10 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distance related bird abundance data (fourth - root transformed) on the 10 year old site. Superimposed clusters from the dendrogram at similarity levels of 8 % - 28 % (dashed line) and 28% - 35% (continuous line) are indicated on the MDS (stress = 0.08).

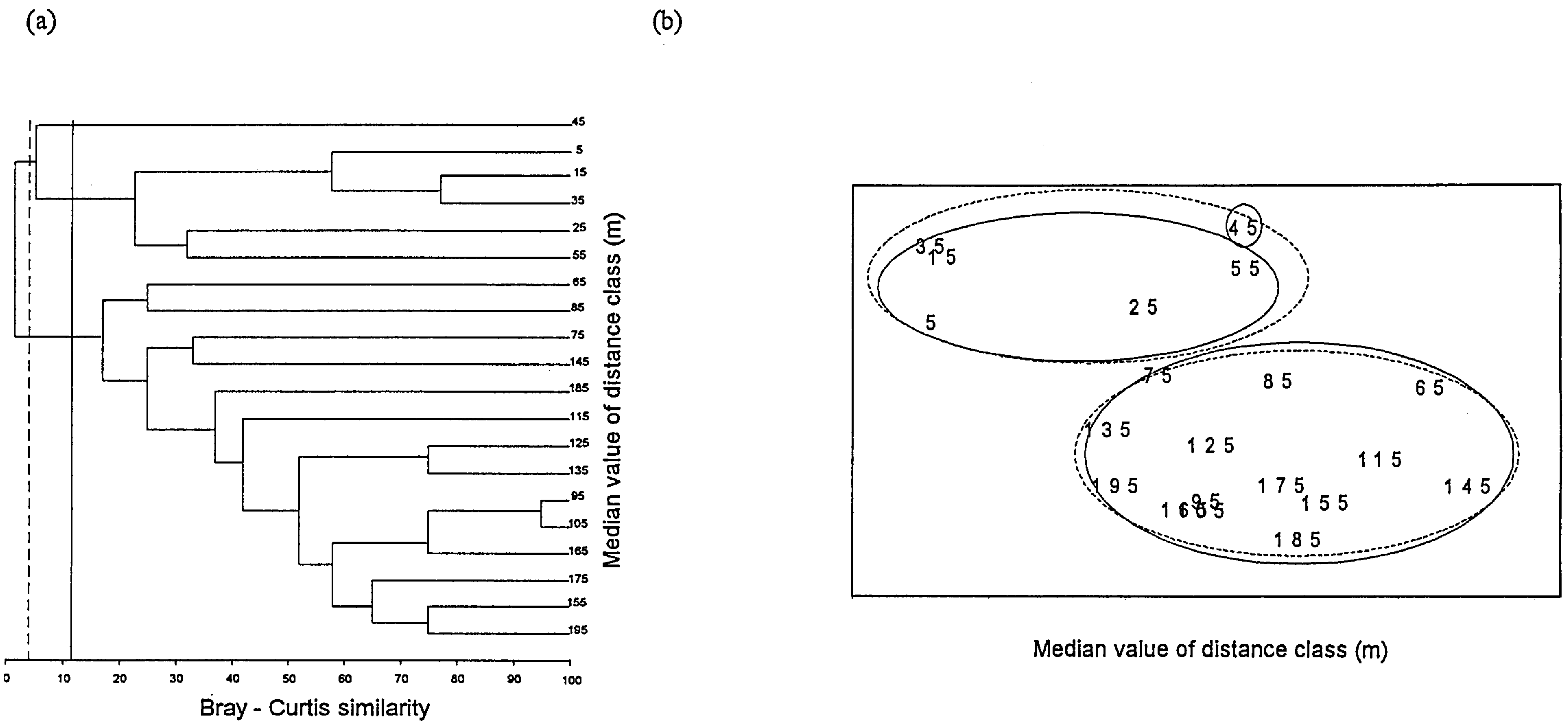

Figure 11 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distances related bird abundance data (fourth - root transformed) on the 14 year old site. Superimposed clusters from the dendrogram at similarity levels of 0% - 5% (dashed line) and 5% - 17% (continuous line) are indicated on the MDS (stress = 0.12).

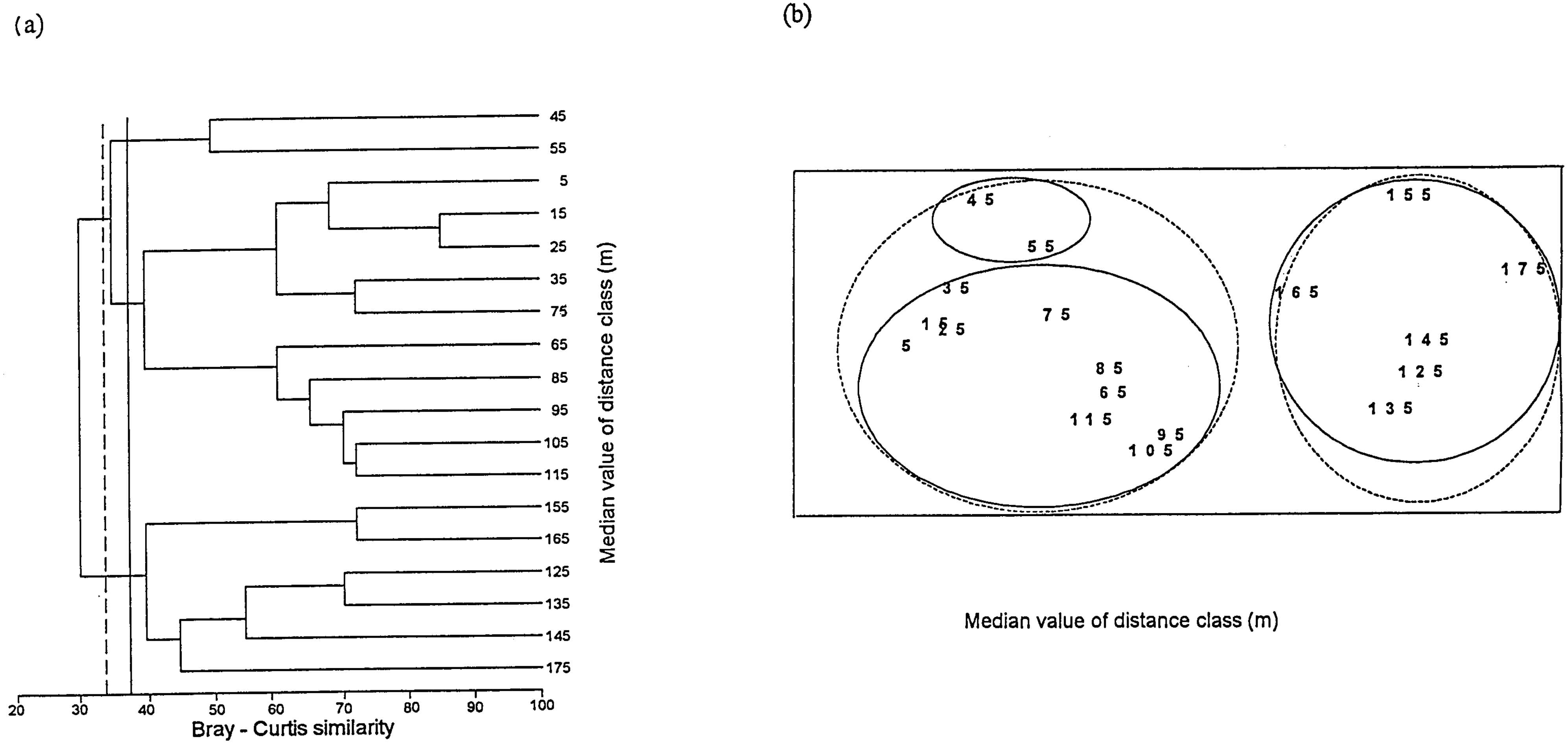

Figure 12 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distances related bird abundance data (fourth - root transformed) of the 18 year old site. Superimposed clusters from the dendrogram at similarity levels of 30% - 35% (dashed line) and 35% - 40% (continuous line) are indicated on the MDS (stress = 0.15).

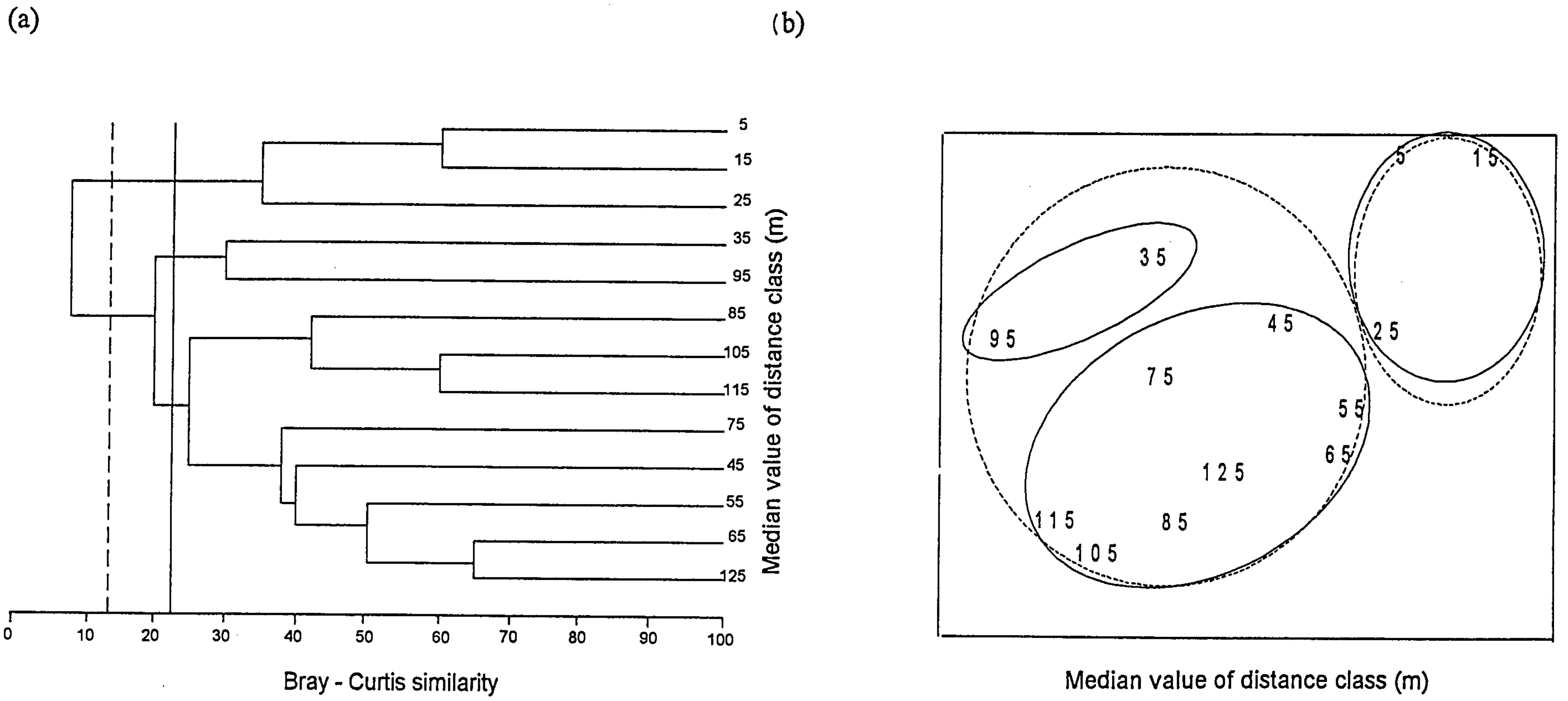

Figure 13 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distance related bird abundance data (fourth - root transformed) of the unmined forest. Superimposed clusters from the dendrogram at similarity levels of 10% - 20% (dashed line) and 20% - 25% (continuous line) are indicated on the MDS (stress = 0.13).

Table 3 Discriminant species of the edge and core assemblages of the different aged sites, calculated from contributions of different species to Bray - Curtis distances in the unmined forest and rehabilitating sites, following Clarke (1993). Calculations were based on fourth - root transformed relative abundances for different species in each site. Species are listed showing their mean contribution (δ_i) to the average distance between assemblages in the different aged sites, with a cut - off point when the cumulative percentage contribution ($\Sigma\delta_i$) is nearest to, and above 50%. The consistency of a species contribution is shown as (δ_i/SD(δ_i)). For scientific names refer to Appendix 2.

Age of site	Species	Mean contribution to edge assemblage δ_i	Mean contribution to core assemblage δ_i	Consistency δ_i/SD(δ_i)	Cumulative percentage contribution $\Sigma\delta_i$
10 years	Olive Sunbird	0.29	0.20	1.47	7.39
	Bleating Warbler	1.14	0.76	1.00	14.33
	Forest Weaver	0.36	0.04	1.67	20.97
	Whitebrowed Robin	0.08	0.11	1.59	27.00
	Cape White-eye	0.39	0.49	1.00	32.66
	Yellowbreasted Apalis	0.48	0.46	0.82	37.96
	Yellow eyed Canary	0.14	0.00	1.26	42.78
	Sombre Bulbul	0.11	0.00	1.24	47.55
	Puffback	0.11	0.02	1.18	52.13
14 years	Puffback	0.55	0.09	1.37	8.68
	Yellowbreasted Apalis	1.13	0.40	1.11	17.07
	Collared Sunbird	0.10	0.00	1.54	24.53
	Cape White-eye	0.35	0.21	1.00	31.45
	Bleating Warbler	2.17	1.07	0.82	37.13
	Natal Robin	0.10	0.00	0.89	42.74
	Dusky Flycatcher	0.25	0.00	0.95	48.32
	Forest Weaver	0.20	0.05	1.00	53.56
18 years	Natal Robin	1.40	0.10	1.23	6.75
	Tawnyflanked Prinia	0.73	0.27	1.17	13.35
	Blackbellied Starling	0.27	0.00	1.29	19.48
	Forktailed Drongo	0.47	0.39	1.20	25.34
	Spotted Flycatcher	0.27	0.04	1.24	30.96
	Olive Sunbird	0.87	0.40	1.05	36.56
	Cape White-eye	1.53	1.28	0.86	42.11
	Forest Weaver	0.53	0.51	1.07	47.44
	Puffback	0.40	0.56	1.07	52.59
Unmined forest	Bleating Warbler	2.00	0.67	0.85	16.55
(unknown age)	Dusky Flycatcher	1.00	0.00	0.84	30.77
	Blackeyed Bulbul	0.00	2.00	0.81	44.45
	Sombre Bulbul	1.00	1.33	0.92	57.20

abundances of species discriminating between the edge and core assemblages for the different sites were similar, though their relative contributions to dissimilarities differed (see Table 3). Also note that for each of the sites the four species contributing most to dissimilarity between edge and core assemblages differed. The species composition varied between edge and core assemblages in all study sites (Table 4).The core assemblage had a higher species richness in all study sites except the unmined forest (Table 4).

Feeding guilds

The number of individuals recorded in the insectivore and nectarivore guild at set distances from the edge decreased significantly with an increase in distance from the edge except for the unmined forest (Figures 14 & 15). In the frugivore guild the 14 year old regenerating and unmined forest did not decrease significantly with an increase in distance from the edge (Figure 16). Significant edge effects existed for all three guilds on the 14 and 18 year old rehabilitating site and for insectivores on the 10 year old site (Table 5). The unmined forest showed no significant edge widths. The significant edge widths calculated for species richness, diversity and number of individuals decreased with increase in regeneration age (Figure 17).

A species list showing the mean abundance values for each species in every aged site for the edge and core assemblages are provided in Appendix 2.

Discussion

Based on the analyses presented here, the bird community of the regenerating coastal dune forests in northern KwaZulu - Natal are influenced by roads traversing these forests. Other studies that described the reaction of birds to an edge without division into functional groups (e.g feeding guilds), showed that species richness decreased from the edge to the forest core (Gates & Gysel 1978; Strelke & Dickson 1980). This coincides with the results from the rehabilitating sites of the present study. In the rehabilitating sites species richness and diversity decreased with an increase in distance from the edge and only in the 10 and 18 year old rehabilitating sites did the number of individuals decrease with an increase in distance from the edge. This tendency probably resulted from the greater number of foliage layers at the edge and from access by birds to more than one habitat type at the clear - cut roads (Kroodsma 1982). The unmined forest showed no relationship

Table 4 The mean (± standard error) species richness, species diversity and number of individuals for edge and core assemblages. The species composition of the edge and core assemblages are listed. The core and edge assemblages were identified as described on p 16 and Figures 10 to 13. Typical species (as identified by Kritzinger 1996) of the 10 and 14 year old rehabilitating and unmined forest have been indicated with asterisks. For scientific names refer to Appendix 2.

Community variables	Study site							
	10 year old rehabilitating site		14 year old rehabilitating site		18 year old rehabilitating site		Unmined forest of unknown age	
	Edge	Core	Edge	Core	Edge	Core	Edge	Core
Species richness	12.3 ± 1.4	3.8 ± 0.8	8.2 ± 1.0	3.5 ± 0.4	11.7 ± 1.0	5.3 ± 1.4	7.7 ± 0.3	7.2 ±1.1
Shannon diversity	2.0 ± 0.09	1.0 ± 0.1	1.7 ± 0.1	0.9 ± 0.1	2.0 ± 0.09	1.5 ± 0.2	1.7 ± 0.5	1.7 ± 0.2
Number of individuals	4.7 ± 0.7	1.9 ± 0.7	4.8 ±1.1	2.5 ± 0.3	11.7 ± 1.5	6.8 ± 1.1	2.8 ± 0.5	2.5 ± 1.3
Total number of species	31	11	23	19	31	17	15	28
Species composition	Blackeyed Bulbul	Bleating Warbler*	African Goshawk	Blackeyed Bulbul*	African Scops Owl	Black Cuckooshrike	Bleating Warbler*	Blackbellied Starling
	Bluegrey Flycatcher	Cape White - eye*	Blackeyed Bulbul*	Bleating Warbler*	Bearded Woodpecker	Bluegrey Flycatcher	Blackeyed Bulbul	Bluegrey Flycatcher
	Black Sunbird	Forest Weaver	Bluegrey Flycatcher	Bluegrey Flycatcher	Blackbellied Starling	Bleating Warbler	Collared Sunbird*	Black Sunbird
	Bleating Warbler*	Forktailed Drongo	Bleating Warbler*	Cape White -eye*	Black Cuckooshrike	Collared Sunbird	Dusky Flycatcher	Blackeyed Bulbul
	Collared Sunbird	Goldenrumped Tinkerbarbet	Collared Sunbird	Forktailed Drongo	Blackeyed Bulbul	Cape White - eye	European Cuckoo	Bleating Warbler*
	Cape White - eye*	Olive Sunbird	Cape White -eye*	Forest Weaver*	Bluegrey Flycatcher	Forest Weaver	Forest Weaver*	Cape White - eye*
	European Cuckoo	Puffback*	Dusky Flycatcher	Olive Sunbird	Bleating Warbler	Goldenrumped Tinkerbarbet	Forktailed Drongo	Collared Sunbird*
	Forktailed Drongo	Spottedbacked Weaver	Forest Weaver*	Puffback*	Cape White - eye	Natal Robin	Natal Robin	Crowned Hornbill
	Fantailed Flycatcher	Whitebrowed Robin	Goldenrumped Tinkerbarbet	Paradise Flycatcher	Collared Sunbird	Olive Sunbird	Olive Sunbird*	Dusky Flycatcher
	Forest Weaver	White-bellied Sunbird	Livingstone Lourie	Pygmy Kingfisher	Cuckoo Hawk	Puffback	Puffback*	Forktailed Drongo
	Goldenrumped Timker barbet	Yellowbreasted Apalis*	Natal Robin	Sombre Bulbul	Dusky Flycatcher	Squaretailed Drongo	Sombre Bulbul	Forest Weaver*
	Grey Sunbird		Olive Sunbird	Squaretailed Drongo*	Forktailed Drongo	Tawnyflanked Prinia	Spotted Flycatcher	Goldenrumped Timker barbet*
	Long billed Crombec		Puffback*	Southern Masked Weaver	Forest Weaver	White-bellied Sunbird	Scarletchested Sunbird	Little Bee - eater
	Natal Robin		Paradise Flycatcher	Tambourine Dove	Goldenrumped Timker barbet	Yellowbreasted Apalis	Tambourine Dove	Longcrested Eagle
	Olive Sunbird		Rattling Cisticola	Tawnyflanked Prinia	Green Twinspot	Yellowbellied Bulbul	White - eared Barbet	Livingstone Lourie
	Paradise Flycatcher		Red - eyed Dove	White-bellied Sunbird	Lizard Buzzard		Yellowbreasted Apalis*	Natal Robin
	Puffback*		Squaretailed Drongo*	Yellowbreasted Apalis*	Natal Robin		Yellow - eyed Canary	Olive Sunbird*
	Pygmy Kingfisher		Tambourine Dove	Yellowbellied Bulbul	Olive Sunbird			Puffback*
	Rudd's Apalis		Tawnyflanked Prinia	Yellow Weaver	Puffback			Sombre Bulbul
	Sombre Bulbul		White-bellied Sunbird		Paradise Flycatcher			Southern Boubou
	Squaretailed Drongo		Yellowbellied Bulbul		Sombre Bulbul			Squaretailed Drongo
	Southern Masked Weaver		Yellowbreasted Apalis*		Scarletchested Sunbird			Thickbilled Weaver
	Spottedbacked Weaver		Yelloweyed Canary		Squaretailed Drongo			Whitebrowed Robin
	Tambourine Dove				Spotted Flycatcher			White - eared Barbet
	Tawnyflanked Prinia				Terrestrial Bulbul			Yellowbreasted Apalis*
	Whitebrowed Robin				Tambourine Dove			Yellowbellied Bulbul*
	White-bellied Sunbird				Tawnyflanked Prinia			Yellow - eyed Canary
	Yellow Weaver				Whitetailed Flycatcher			Yellow Weaver
	Yellowbreasted Apalis*				Yellowbreasted Apalis			
	Yellowbellied Bulbul				Yellowbellied Bulbul			
	Yellowspotted Nicator				Yelloweyed Canary			

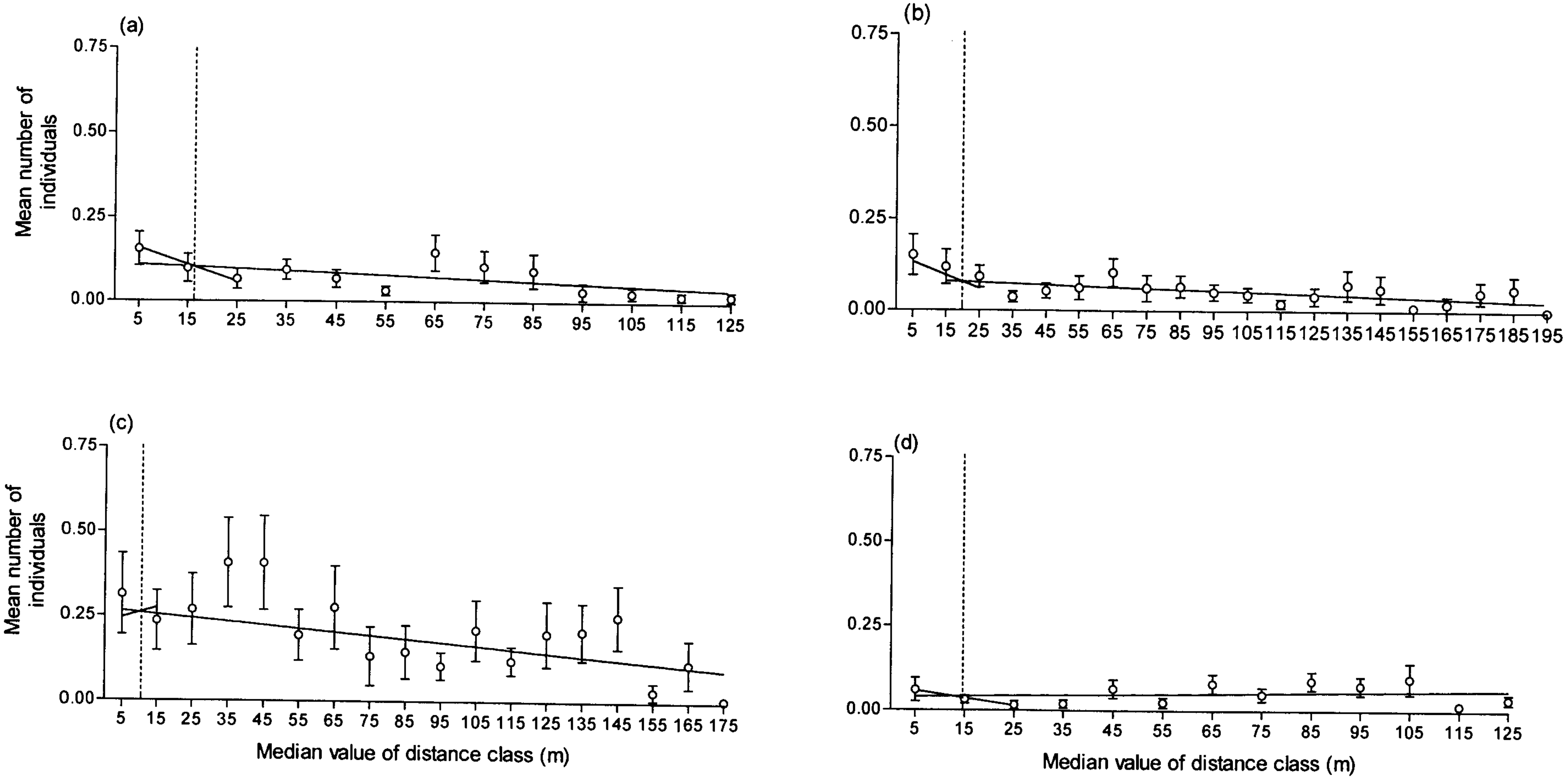

Figure 14 Mean (± standard error) number of individuals of the insectivore guild of the rehabilitating and unmined coastal dune sites: (a) 10 year old site, (b) 14 year old site, (c) 18 year old site and (d) and unmined forest. The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (Table 5). The regression equations are listed in Appendix 1.

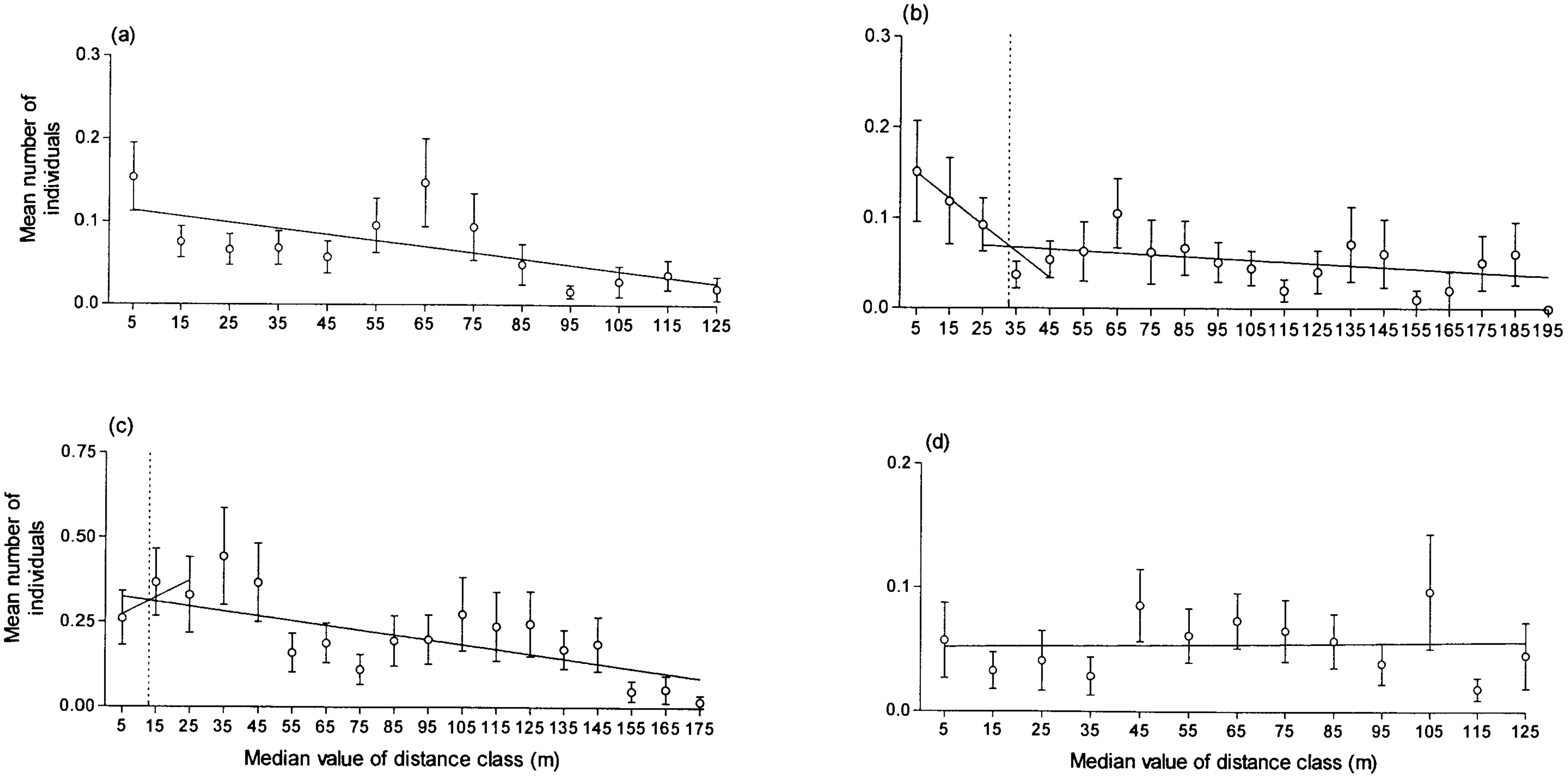

Figure 15 Mean (± standard error) number of individuals of the nectarivore guild of the rehabilitating and unmined coastal dune sites: (a) 10 year old site, (b) 14 year old site, (c) 18 year old site and (d) and unmined forest. The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (Table 5). The regression equations are listed in Appendix 1.

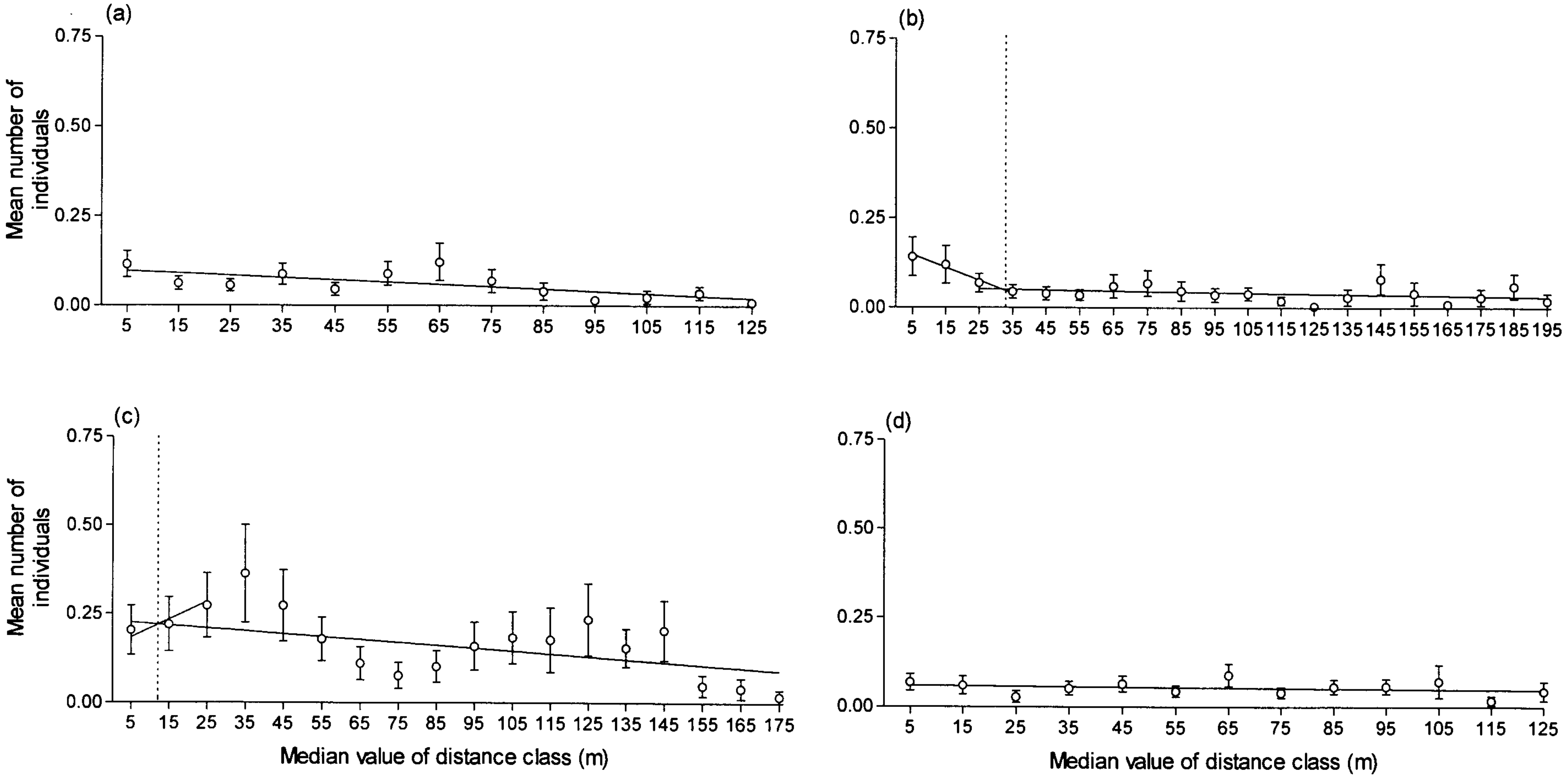

Figure 16 Mean (± standard error) number of individuals of the frugivore guild of the rehabilitating and unmined coastal dune sites: (a) 10 year old site, (b) 14 year old site, (c) 18 year old site and (d) and unmined forest. The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (Table 5). The regression equations are listed in Appendix 1.

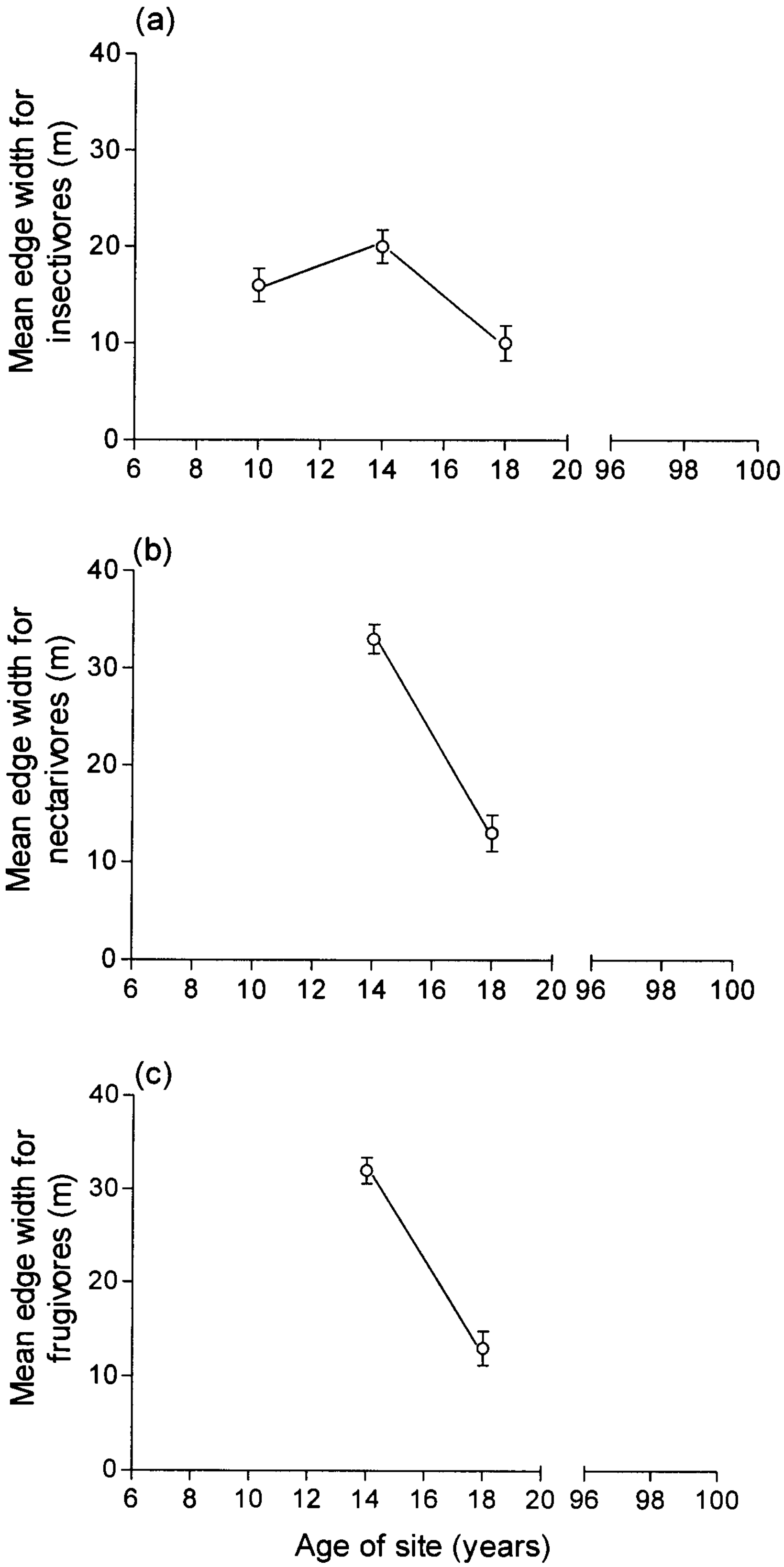

Figure 17 Mean (± standard error) edge widths for feeding guilds (as described on p 16 and presented in Table 5) of the bird community where (a) represents the insectivores, (b) the nectarivores and (c) the frugivores. In this figure the unmined forest is presented as 100 years old.

Table 5 Mean (± standard error) edge widths in metres (see text p 16) for insectivores, nectarivores and frugivores in each study site, followed by the *F* and *p* - values.

Feeding guild	Study site											
	10 year old rehabilitating site			14 year old rehabilitating site			18 year old rehabilitating site			Unmined forest of unknown age		
	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*
Insectivores	16 ± 1.7	5.27	<0.01	20 ± 1.7	10.67	<0.001	10 ± 2.8	6.18	<0.02	-	6.19	ns
Nectarivores	-	1.13	ns	33 ± 1.5	7.10	<0.01	13 ± 1.9	6.15	<0.02	-	1.58	ns
Frugivores	-	1.72	ns	32 ± 2.4	16.97	<0.0001	13 ± 1.8	5.24	<0.02	-	1.07	ns

of species richness, species diversity or number of individuals with an increase in distance from the edge. In the unmined forest the vegetation is dense at all distances from the road. Thus the roads may not influence the bird assemblages enough to cause a trend in richness, diversity or number of individuals. In a study conducted elsewhere in an undisturbed forest in KwaZulu - Natal, Krüger & Lawes (1997) could not illustrate a difference in species richness with increasing distance from the edge.

Changes in microclimate and the distribution of suitable habitats can affect the dispersal abilities and home range size of birds and therefore influence the distribution of birds from forest edge to forest core (Wiens 1989). Inter study variability in the effect of edges (e.g. Kroodsma 1982), may be explained by differences in the physiognomy (Matlack 1993; Matlack 1994), orientation (Brothers & Spingarn 1992; Fraver 1994; Matlack 1994), level of disturbance (Fox *et al.* 1997), habitat type surrounding edges, age of edges (Saunders *et al.* 1991; Matlack 1994) and abundance and type of predator occurring at the edge (Gates & Gysel 1978). There is strong evidence that more predators occur in the edge zone (Gates & Gysel 1978; Marini *et al.* 1995), which may cause an increase in nest predation with decreasing distance from the edge (Gates & Gysel 1978; Yahner & Wright 1985; Ratti & Reese 1988; Paton 1994; Marini *et al.* 1995). Adding to the risk of higher predator densities, forest core species that occur near the edges after fragmentation may be poorly adapted to predation near artificial edges and thus become easy prey (Ratti & Reese 1988; Marini *et al.* 1995).

In the present study the calculated edge width for species richness, diversity and number of individuals decreased with an increase in regeneration age in the 14 and 18 year old rehabilitating sites and unmined forest. Narrower edge widths found in the older sites (such as the unmined forest), has dense vegetation acting as a physical and visual obstruction for both birds and their predators (La Rue *et al.* 1994). In the 10 year old rehabilitating site no significant edge widths were calculated. This site has a relative homogeneous vegetation structure, which may explain the absence of an edge width.

Several studies showed that birds in forests can be separated into two assemblages namely, an edge and core assemblage (Johnston 1947; Kroodsma 1982; Andrén 1994). In order to distinguish between the edge and core species various methods have been applied, such as defining edge species as the species reaching their highest density within 60 m's from the edge, and core species as the species reaching their highest density beyond 60 m's from the edge (Kroodsma 1982). In the present study an edge and core assemblage was identified in all the different aged sites by using multi - variate analyses techniques (Figures 10 - 13).

The distinction between the edge and core assemblages in the rehabilitating sites can be ascribed to differences in species richness, diversity and number of individuals in the rehabilitating sites (Table 4). Some birds which prefer occurring near the edge in the regenerating and unmined coastal dune forests of the study area, typically occur in mixed habitat types, or forage in the undergrowth (see Maclean 1993; Harrison, Allan, Underhill, Herremans, Tree, Parker & Brown 1997). They are thus ideally adapted for occurring in the edge zone. These birds occurring on the edge include species such as the Blackeyed Bulbul (*Pycnonotus barbatus*), Tambourine Dove (*Turtur tympanistria*), Collared Sunbird (*Anthreptes collaris*) (see Table 4). They also included species contributing significantly to the average dissimilarity between the edge and core assemblages (Table 3). Species such as the Yelloweyed Canary (*Serinus mozambicus*) occurred only in the edge zone in the rehabilitating sites, but in both the edge and core assemblages in the unmined forest. Some species occurred only in the edge zone of the 10 and 14 year old rehabilitating sites, which included the Natal Robin (*Cossypha natalensis*), Bluegrey Flycatcher (*Muscicapa caerulescens*) and Paradise Flycatcher (*Terpsiphone viridis*).The species contributing significantly to both the edge and core assemblages differed between age of regenerating site (see Table 3). In the 10 year old site, the typical species of the site (as identified by Kritzinger 1996) occurred in the edge and core assemblages of the 10 and 14 year old rehabilitating sites. Three species (Cape White - eye (*Zosterops pallidus*), Goldenrumped Tinkerbarbet (*Pogoniulus bilineatus*) and Yellowbellied Bulbul (*Chlorocichla flaviventris*)) only occurred in the core assemblage and not in the edge assemblage of the unmined forest (see Table 4). This type of comparison could not be done with the data for the 18 year old site due to the regeneration age of the sites not having reached 18 years when Kritzinger (1996) collected data. Only a few species such as the Olive Sunbird (*Nectarinia olivacea*), Puffback (*Dryoscopus cubla*), Natal Robin (*Cossypha natalensis*) and Bleating Warbler (*Camaroptera brachyura*), which are usually in a bird party and are characteristic of mixed habitats, occurred in the edge and core assemblages in all the study sites (Table 3).

Gates & Gysel (1978) showed that bird species nesting near edges had characteristics of mixed breeding habitats and thus that they are adapted to the changing environment and neighbouring habitats at edges (Andrén 1994).

Although an edge and core assemblage was identified for the 10 year old regenerating site, there were no species occurring only in the core assemblage. This coincides with the results of the absence of the edge width in the community variables of the 10 year old regenerating site, which may result from the homogeneous vegetation structure of the site (Table 4). In a newly fragmented area species that usually occurred in the forest core will be invaded by edge species when that new edge forces those edge species to move inwards towards the core (Gates & Giffen 1991). It is thus not surprising that no species could be identified as only a core species in the rehabilitating sites. Species such as the Bleating warbler (*Camaroptera brachyura*), Olive Sunbird (*Nectarinia olivacea*), Puffback (*Dryoscopus cubla*) and Yellowbreasted Apalis (*Apalis flavida)* occurred throughout all the study sites in both the edge and core assemblages (see Table 4). These species are usually part of bird parties foraging in the upper and middle strata (Maclean 1993). The changes in species composition of the edge and core assemblages between different aged sites may be due to edge species occurring along the edge, core species occurring more frequently in the forest core and an apparent tendency for unaffected forest bird species to occur along the edge (see Kroodsma 1982). This tendency for unaffected species to occur along the edge may not necessarily be due to the attractiveness near the edge, but may rather be due to the open road that form a habitat border beyond which the forest species do not extend their range (Kroodsma 1982).

To further understand the impact of edges on bird communities, one must consider the ecological processes mediated through plant - animal interactions (Restrepo & Gómez 1998). An edge may influence the behaviour and distribution of seed dispersers, thereby influencing the distribution of plants along edges. Differences among feeding guilds suggest that ecological interactions mediated through plant - animal interactions may not equally affect edges (Restrepo & Gómez 1998). Insects often occur more frequently near edges than further into the forest (Cappucino & Martin 1997; Courtney & Courtney 1982), probably due to the increased amount of sunlight at the edge compared with the forest core causing an increase in the nutritional value of leaves (Collinga & Louda 1989). This could lead to an increase in birds near edges (Strelke & Dickson 1980). This phenomenon is shown in the present study. Insectivores and nectarivores of the rehabilitating sites decreased with an increase in distance from the edge. Frugivores only decreased from the edge towards the forest core in the 10 and 18 year old rehabilitating sites. It has been shown that the edge effect of frugivores, nectarivores and insectivores coincide with the occurrence of their resource bases (Hansson 1998; Noss 1983; Restrepo & Gómez 1998).The significant edge width calculated for frugivores and nectarivores in the 14 and 18 year old rehabilitating sites may be related to the mixed habitat types occurring at the edge. And in the unmined forest due to the thick vegetation throughout the forest, the feeding guilds showed no response to an increase in distance from the edge. The presence or absence of undergrowth and big, old trees in the study sites influence the species composition of the bird communities of the rehabilitating sites and unmined forest (Kritzinger 1996), thereby also influencing the effect that edges would have on the community.

In the present study the results of the two methods used to estimate the influence of edges on the bird community as a whole (quantifying the edge width and distance where a division into an edge and core assemblage occurred) confounded each other on the extent of the edge effect, but both confirmed that edges have an influence on the bird community in these rehabilitating coastal dune forests. These differences may be due to the community characteristics (e.g. species richness) analysed and interpreted separately, only to partly portray the influence that edges have on the bird community. By using multi - variate analyses the community attributes, such as species composition and species abundance is taken into account at the same time. Due to a high species composition in the bird community, with birds showing various types of breeding, nesting and feeding, the community characteristics analysed separately may be expected to show different results than results of analyses interpreted at the same time.

Chapter 4

Edge effects in the millipede communities

Introduction

Millipedes occur commonly in the study area (see Van Aarde *et al.* 1996b) and based on studies conducted elsewhere are known to play an important role in the fragmentation of litter by making it susceptible for litter decay (Swift, Heal & Anderson 1979). Factors affecting the local distribution of millipedes, such as humidity and temperature (Swift, *et al.* 1979) may have consequences for the recovery of ecosystem function as a restoration goal, especially through the influence that artificial edges, such as these resulting from roads, may have on these environmental variables.

Millipede communities in the rehabilitating coastal dune forests in northern KwaZulu - Natal follow the expected trends for successional development of a community (Van Aarde *et al.* 1996b). During the early stages of rehabilitation (9 - 11 years of age) species replacement is seen to some extent, with *Centrobolus fulgidus* acting as a pioneer species and *Centrobolus richardi* dominating in the older sites (12 - 14 years of age) (Van Aarde *et al.* 1996b). However, *C. richardi* was not the first species to colonize the rehabilitating areas after mining, but *Spinotarsus anguliferus* (Van Aarde *et al.* 1996b). The trends in species numbers and density of some early colonizing species increasing to a peak, to decline again (Van Aarde *et al.* 1996b) agreed with the study on millipedes of Dunger & Voigtländer (1992) on reclaimed land. In the study by Van Aarde *et al.* (1996b) the millipede species composition of the unmined forest differed from these of rehabilitating sites, predominantly due to the absence and/or rarity of some forest specialists, such as *Ulodesmus* sp. (Van Aarde *et al.* 1996b). In the study area millipede diversity increases, but density decreases, with an increase in coastal dune regeneration age (Van Aarde *et al.* 1996b).

The present chapter is directed at assessing the influence of roads as edges on the millipede community variables of different ages.

Materials and methods

Data collection

Surveys on the millipede assemblages were conducted from 28 October to 28 November 1997 as described by Van Aarde, *et al.* (1996b). Quadrats were sampled from dawn to 11h00 for millipedes occurring on the ground (litter strata), occurring from the ground to 30 cm (shrub strata) and for those occurring between 30 cm and three metres above the ground (tree strata). Individuals seen were identified against reference material kept at the Natural Science Museum in Durban, South Africa.

Data analyses

Distance - specific densities were calculated as number of individuals per m^2 and are presented as mean values for each species, followed by one standard error of the mean for each of the distance classes. Community variables and edge widths were calculated and community analyses conducted as described in Chapter 2 (p 16). Percentage contribution and consistency of contribution of millipede species to distinguish between edge and core assemblages were calculated using the method SIMPER as described in Chapter 2 (p 16). Species richness, diversity, number of individuals and species composition were calculated for the edge and core assemblages.

Contribution of Centrobolus fulgidus *to different strata in relation to distance from edge*

To establish the influence of edge effects with change in regenerating age and strata, the distribution of *C. fulgidus*, was studied. The three strata sampled were the litter, shrub and tree strata.

Results

Community variables

Based on slopes of least square regression lines used to calculate edge widths, the species diversity, richness and number of individuals of the millipede community did not vary significantly (p <0.05) with increase in distance from edge in rehabilitating and unmined coastal dune forests (Figures 18 - 20).

Quantifying the edge widths

A significant edge width for species richness existed at 12 m in the 10 year old regenerating site (F = 43.3 m, p <0.02) (Table 6). In the 14 year old regenerating site, a significant edge width was identified for Shannon diversity and number of individuals (Table 6). In the 18 year old site no edge width could be identified for any of the community variables. In the unmined forest all three variables had significant edge widths (Table 6). Due to the edge widths for any one of the variables not being significant in two regeneration aged sites following each other, no trend could be distinguished for this analyses (Table 6). The regression equations for the edge width calculations are listed in Appendix 3.

The community in space

Cluster analyses of millipede community data for the 10 year old site distinguished between an edge and core assemblage, with the former extending to 20 m, and the latter ranging from 40 m to 80 m. However, here the core assemblage included the 0 metre interval (stress = 0.01) (Figure 22). On the 14 year old site, the edge assemblage extended to 10 m, and the core assemblage from 20 m to 100 m (stress <0.0001) (Figure 23). On the 18 year old site the edge assemblage extended from 0 metres to 20 m, while the core assemblage ranged from 40 m to 120 m from the edge (stress = 0.04) (Figure 24). In the unmined forest the edge assemblage extended from 0 metres to 40 m from the road, while the core assemblage extended from 60 m to 80 m (stress = 0.01) (Figure 25).

The analysis of similarity based on edge and core assemblages identified with cluster analysis showed significant differences between these assemblages in the 10 year old site (R = 0.75, p = 0.067), in the 14 year old site (R = 1.00, p = 0.048), in the 18 year old site (R = 0.92, p = 0.018) and unmined forest (R = 0.61, p = 0.051). Therefore in all the study sites the differences between edge and core assemblages were larger than within these assemblages.

Species analyses

The mean percentage contribution of each species, either to the edge or the core assemblages (as

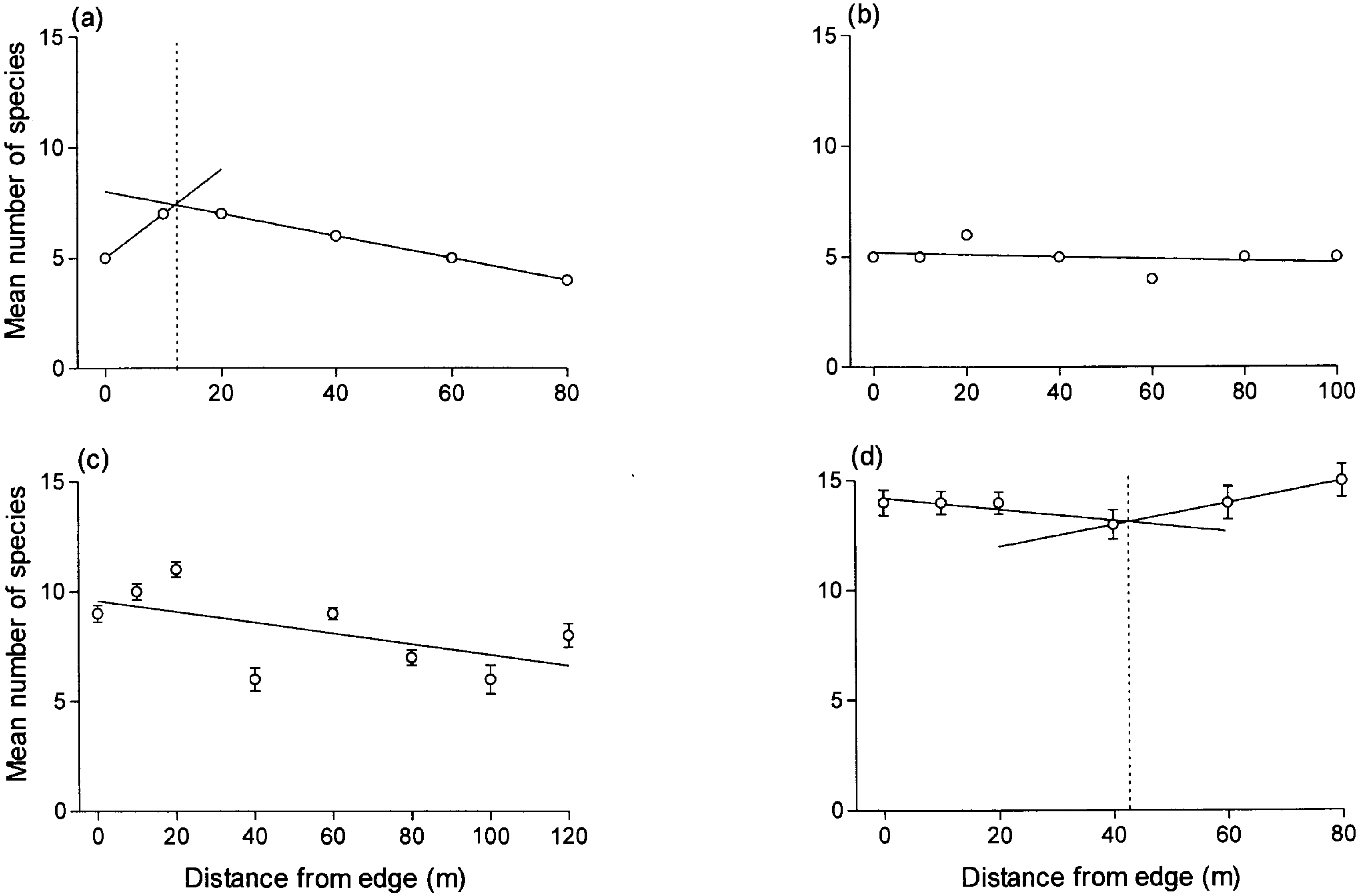

Figure 18 Mean (± standard error) species richness for millipedes in regenerating and unmined coastal dune forests: 10 year old site (a), 14 year old site (b), 18 year old site (c) and the unmined forest (d). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 6). The regression equations are listed in Appendix 3. The standard errors of the 10 and 14 year old rehabilitating sites are very small.

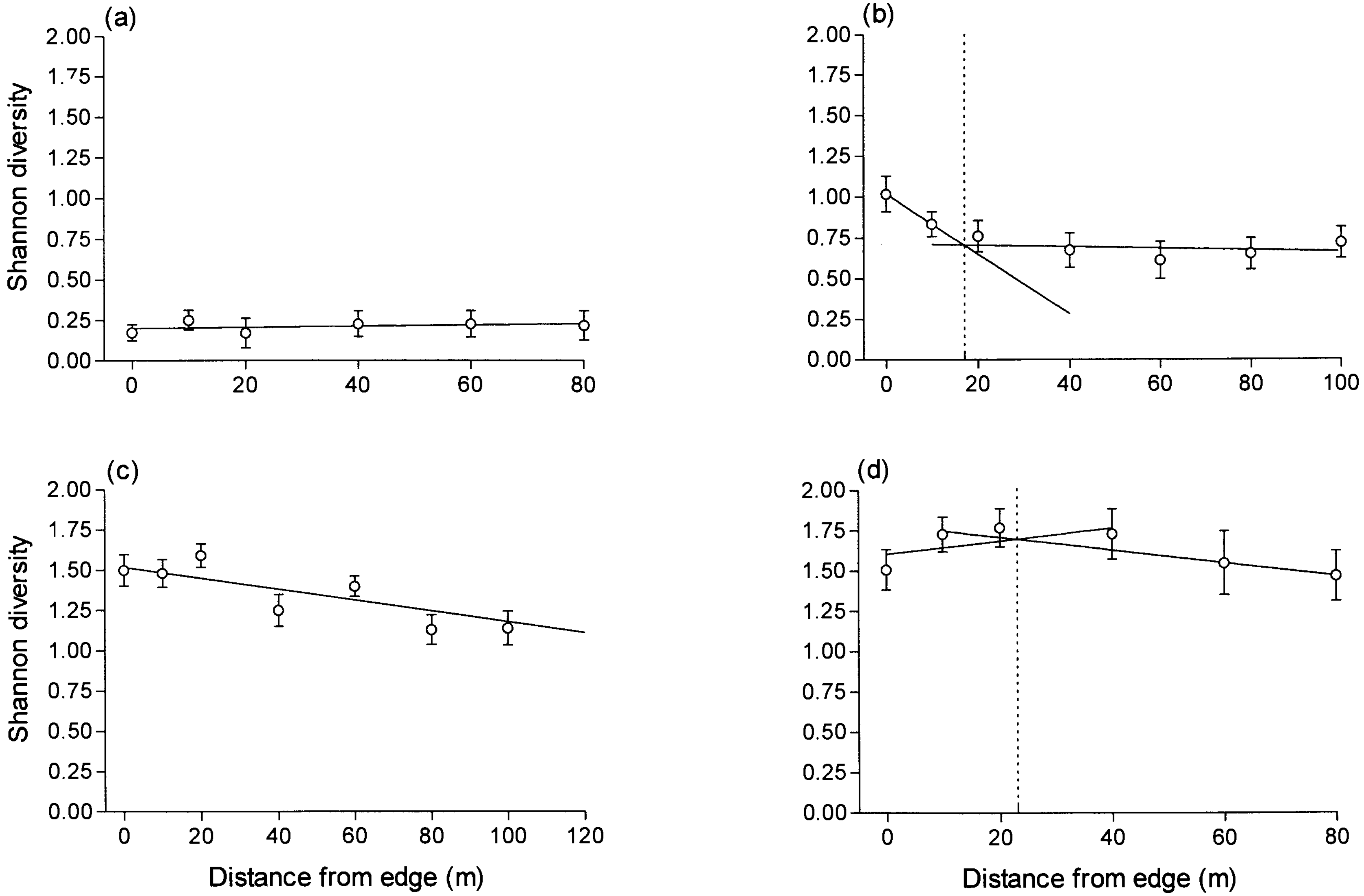

Figure 19 Mean (± standard error) Shannon diversity for millipedes in regenerating and unmined coastal dune forests: 10 year old site (a), 14 year old site (b), 18 year old site (c) and the unmined forest (d). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 6). The regression equations are listed in Appendix 3.

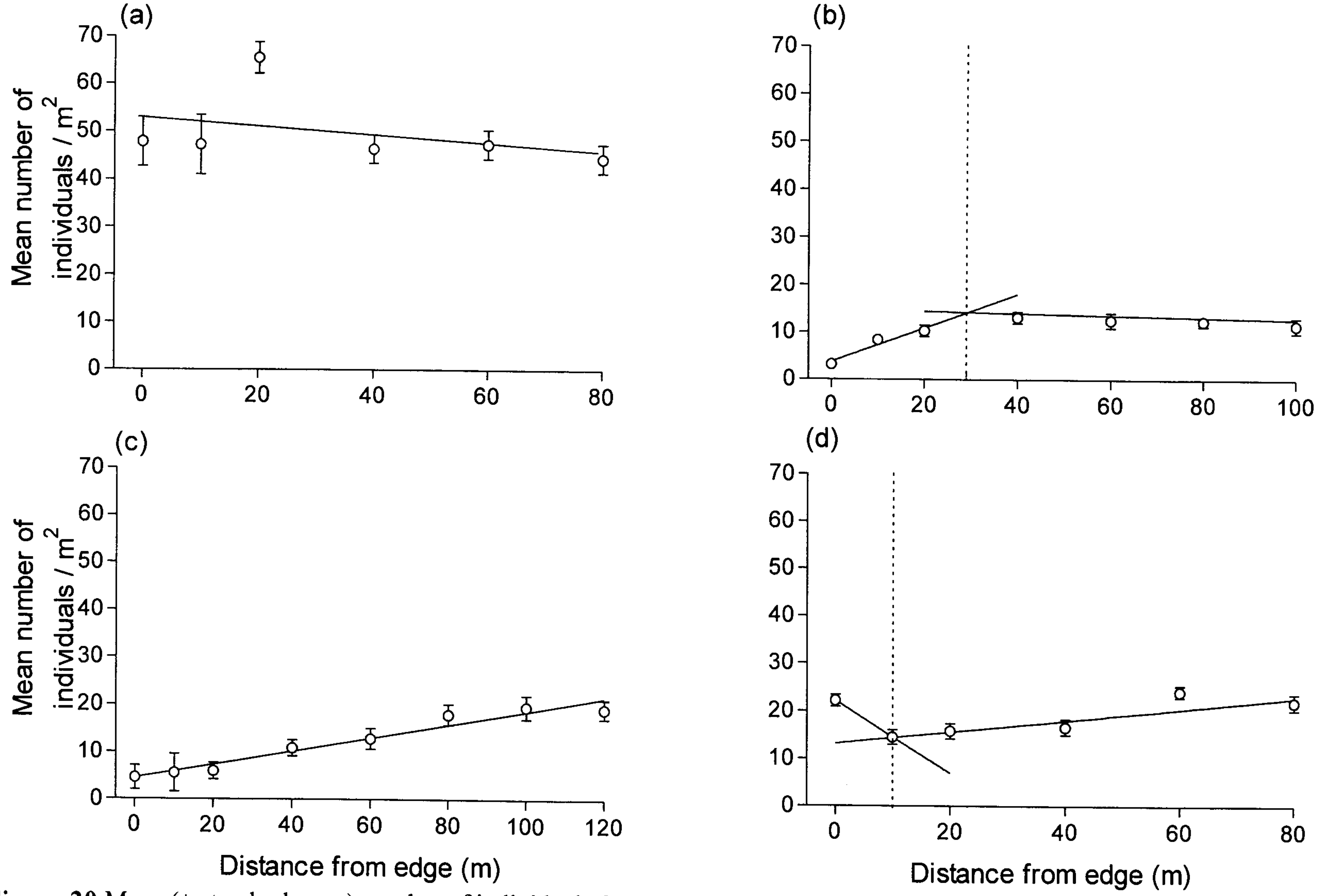

Figure 20 Mean (± standard error) number of individuals for millipedes in regenerating and unmined coastal dune forests: 10 year old site (a), 14 year old site (b), 18 year old site (c) and the unmined forest (d). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 6). The regression equations are listed in Appendix 3.

Table 6 Mean (± standard error) edge width in metres (see text p 16) for species richness, species diversity and number of individuals for milliped in each study site, followed by the calculated *F* and *p* - values.

Community variable	Study site											
	10 year old rehabilitating site			14 year old rehabilitating site			18 year old rehabilitating site			Unmined forest of unknown a		
	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	
Species richness	12 ± 1.3	43.2	< 0.02	-	0.0	ns	-	0.5	ns	42 ± 1.4	8.0	<
Species diversity	-	2.3	ns	16 ± 2.5	27.2	< 0.01	-	0.3	ns	22 ± 1.6	23.9	<
Number of individuals	-	1.8	ns	27 ± 1.7	69.4	<0.001	-	1.2	ns	10 ± 1.7	4.4	<

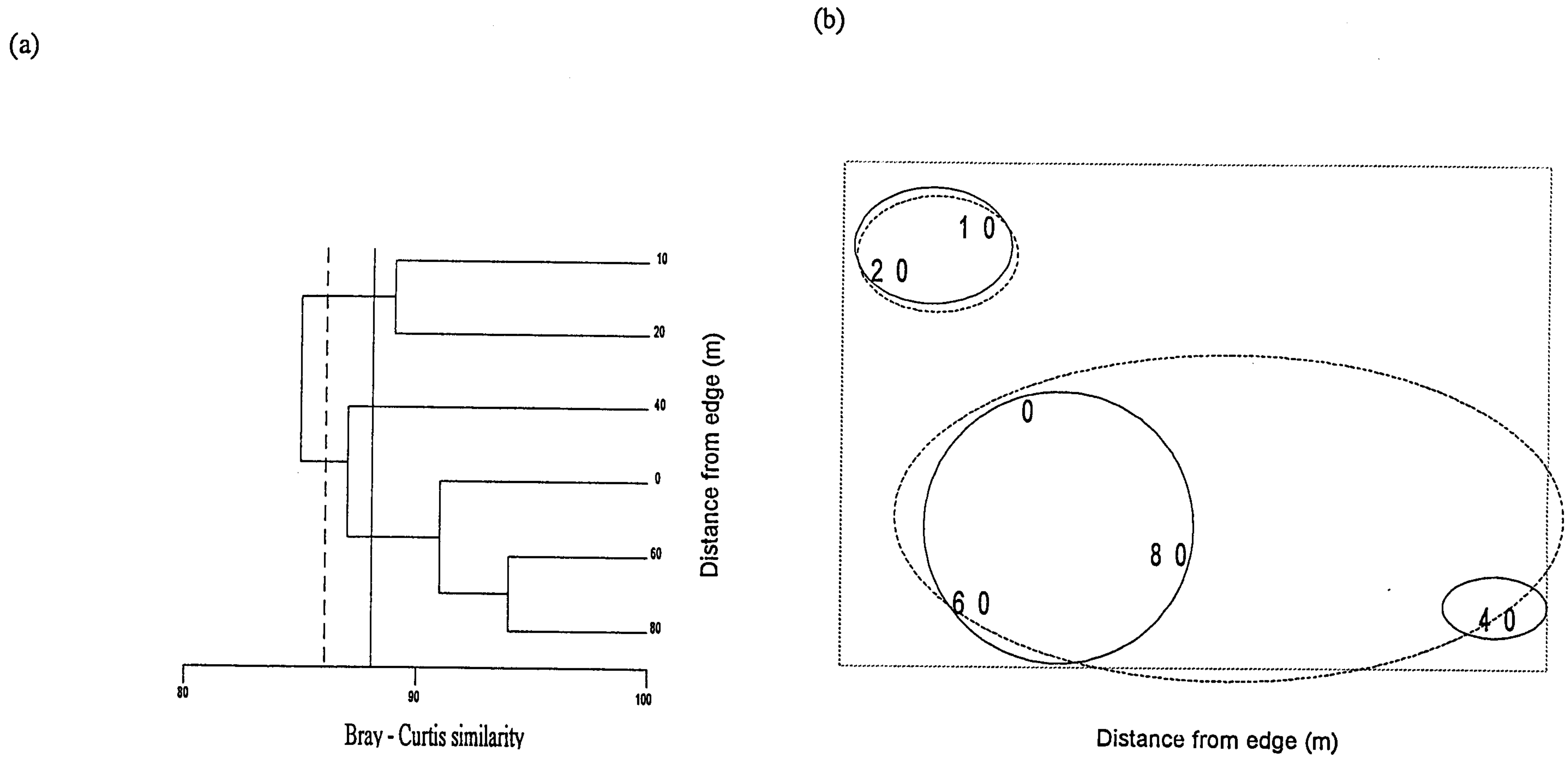

Figure 21 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distance related millipede data (fourth - root transformed) in the 10 year old site. Superimposed clusters from the dendrogram at similarity levels of 85% - 87% (dashed line) and 87% - 89% (continuous line) are indicated on the MDS (stress = 0.01).

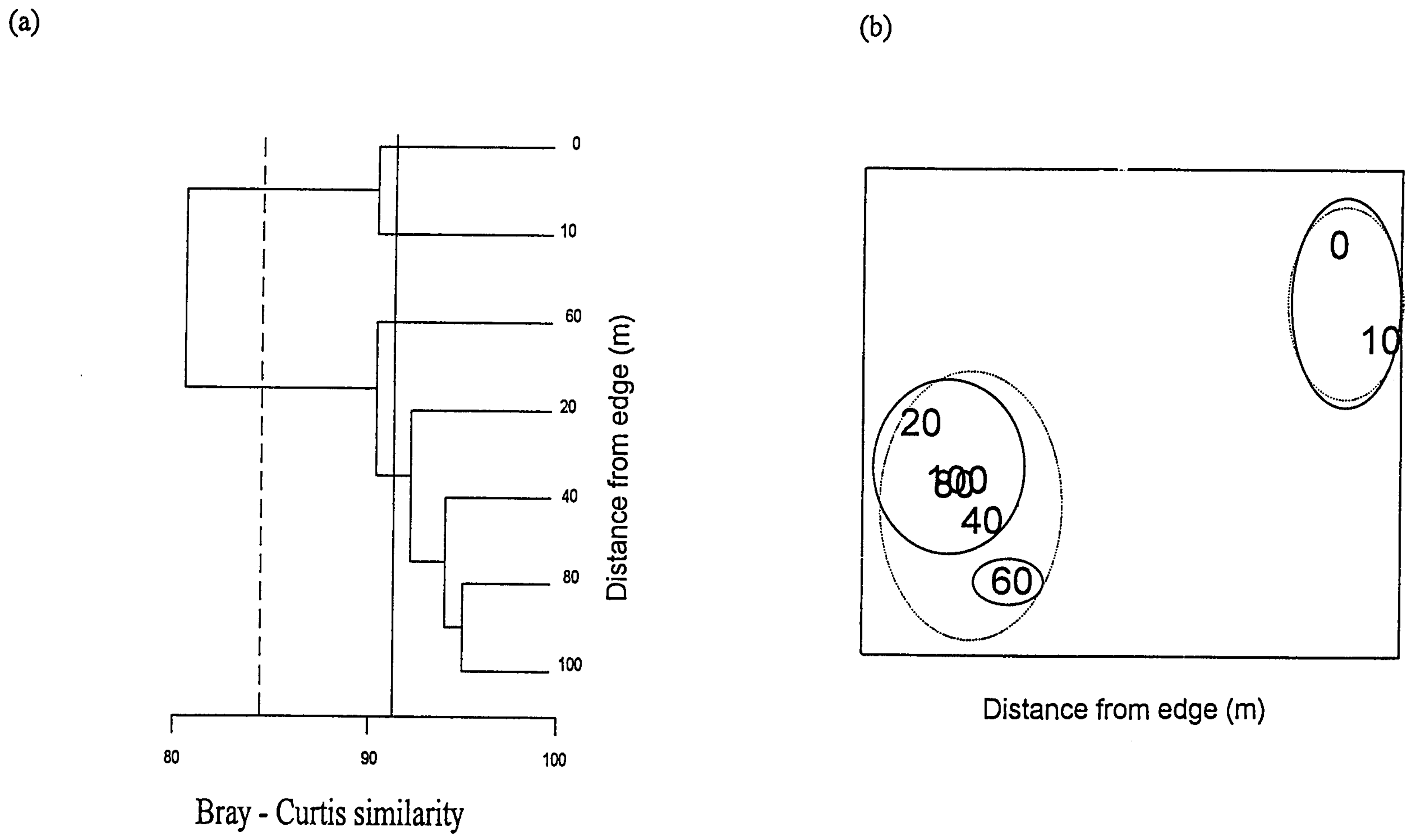

Figure 22 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distance related millipede data (fourth - root transformed) in the 14 year old site. Superimposed clusters from the dendrogram at similarity levels of 78% - 90% (dashed line) and 90% - 93% (continuous line) are indicated on the MDS (stress < 0.0001).

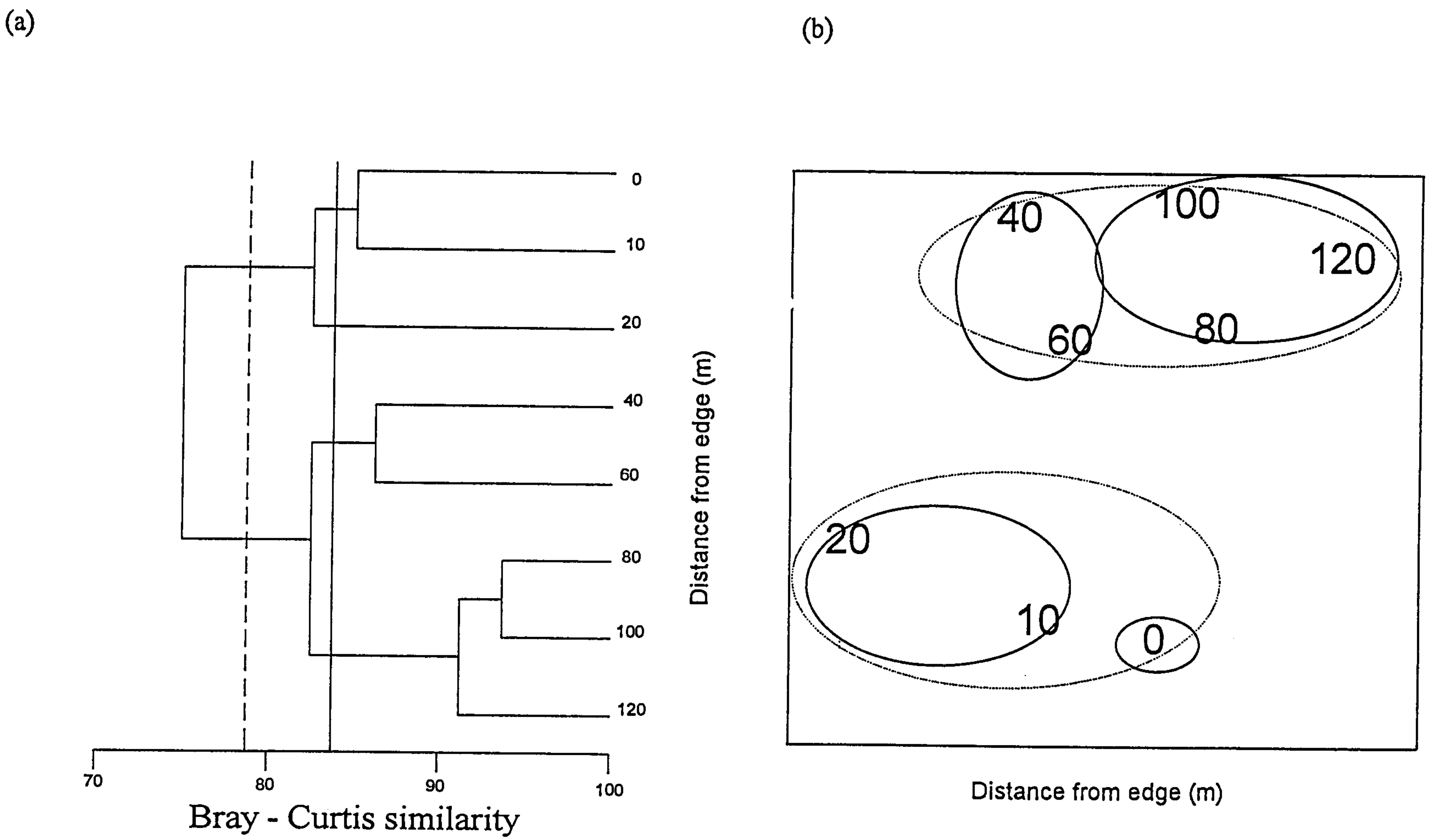

Figure 23 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distance related millipede data (fourth - root transformed) in the 18 year old site. Superimposed clusters from the dendrogram at similarity levels of 75% - 83% (dashed line) and 83% - 92% (continuous line) are indicated on the MDS (stress < 0.0001).

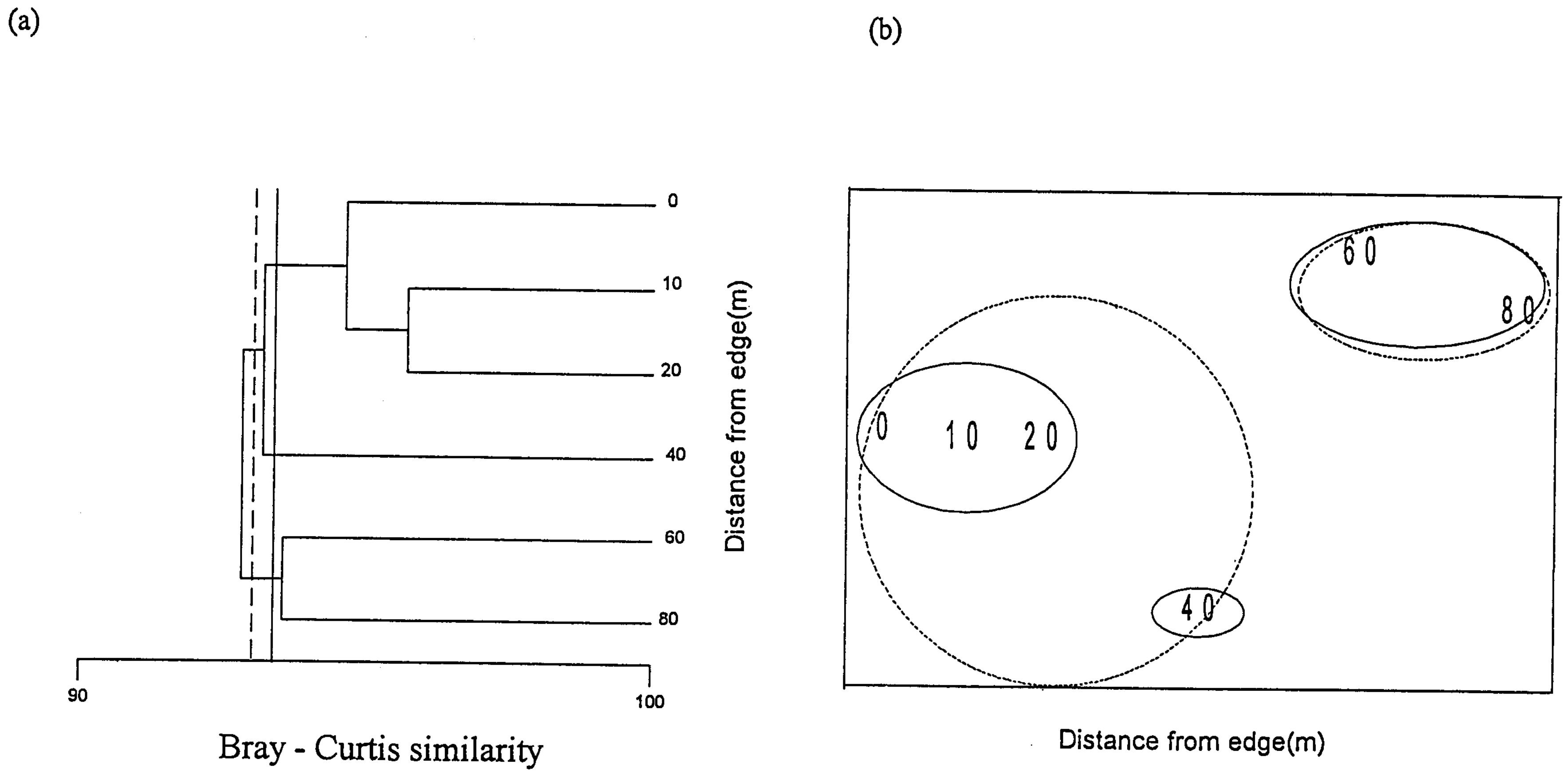

Figure 24 Dendrogram (a) and multi - dimensional scaling ordination (MDS) in two - dimensions (b) of the distance related millipede data (fourth - root transformed) of the unmined forest. Superimposed clusters from the dendrogram at similarity levels of 93% - 93.5% (dashed line) and 93.5% - 94% (continuous line) are indicated on the MDS (stress = 0.04).

identified by cluster analyses and multi - dimensional scaling) as well as the consistency of each species to contribute to the dissimilarity between the edge and core assemblages was calculated as described on p 17. The species contributing to a cumulative 50% of the total mean percentage contribution to dissimilarity between the edge and core assemblages are listed in Table 7. The species contributing to 50% cumulative contribution of dissimilarity between the edge and core assemblages in the 18 year old site, were *Centrobolus fulgidus, Orthoporoides* sp., *Spinotarsus* sp. and *Juliaformia* sp. 2. In the unmined forest, five species contributed to the 50% level of cumulative contributions of dissimilarity between the edge and core assemblages. They were *C. fulgidus, Sphaerotherium punctulatum, Ulodesmus* sp., *Gnomeskelus tuberosus microdens* and *Orthoporoides* sp.

In the regenerating sites *Orthoporoides* sp. only occurred in the edge assemblage, but in the unmined forest, this species occurred in both assemblages (Table 8). Species such as *C. fulgidus, C. richardi, G. tuberosus microdens* and *Spinotarsus* sp. occurred throughout all the sites in both the edge and core assemblages (Table 8).

Contribution of Centrobolus fulgidus *to different strata in relation to distance from edge*

No significant edge widths were identified for *C. fulgidus* in either of the strata of the 10 year old rehabilitating site (Table 9 and Figure 25), but in the 14 and 18 year old rehabilitating site all three strata showed a significant edge width (Table 9 and Figures 26 & 27). In the unmined forest only the litter and shrub strata showed a significant edge width (Table 9 and Figure 28). The trends in calculated edge width with increase in regeneration age is shown in Figure 29.

A species list showing the mean densities for millipede species in the edge and core assemblages of the study sites is given in Appendix 4.

Discussion

Millipedes are important in the fragmentation of accumulated leaf matter (Wallwork 1976) and facilitate microbial decomposition (Swift *et al.* 1979). They usually occur in areas with high moisture and damp soil to ensure permeability for burrowing (Lawrence 1984) and are therefore susceptible to changes in humidity and temperature. Based on the present analyses, the millipede community in regenerating coastal dune forests north of Richards Bay is influenced by roads

Table 7 Discriminant species of the edge and core assemblages in the different aged sites, calculated from contributions of different species to Bray - Curtis distances in the unmined forest and rehabilitating sites, following Clarke (1993). Calculations were based on fourth - root transformed relative densities for different species in each site. Species are listed in order of their mean contribution (δ_i) to the average distance in the different aged sites, with a cut - off point when the cumulative percentage contribution ($\Sigma\delta_i$) is nearest to and above 50%. The consistency of a species contribution is shown as ($\delta_i/SD(\delta_i)$).

Age of study site	Species	Mean contribution to edge assemblage δ_i	Mean contribution to core assemblage δ_i	Consistency $\delta_i/SD(\delta_i)$	Cumulative percentage contribution $\Sigma\delta_i$
10 years	*Doratogonus* sp.	0.03	0.00	10.24	26.58
	Juliaformia sp. 3	0.02	0.01	1.62	44.75
	Orthoporoides sp.	0.01	0.00	0.93	56.92
14 years	*Doratogonus* sp.	0.00	0.07	7.87	26.15
	Orthoporoides sp.	0.03	0.00	17.06	48.26
	Centrobolus fulgidus	3.87	9.10	1.67	68.28
18 years	*Centrobolus fulgidus*	1.81	7.53	2.62	14.76
	Orthoporoides sp.	0.05	0.00	5.94	28.51
	Spinotarsus sp.	1.39	5.31	3.64	40.96
	Juliaformia sp. 2	0.01	0.06	1.27	50.88
Unknown age (unmined forest)	*Centrobolus fulgidus*	8.47	13.30	2.01	12.53
	Sphaerotherium punctulatum	0.03	0.08	1.00	24.62
	Ulodesmus sp.	0.00	0.01	0.94	36.04
	Gnomeskelus tuberosus microdens	0.19	0.46	5.21	45.36
	Orthoporoides sp.	2.68	3.60	1.48	53.98

Table 8 The mean (± standard error) species richness, species diversity and number of individuals for edge and core assemblages. The species composition of the edge and core assemblages are listed. The core and edge assemblages were identified as described on p 16 and in Figures 22 - 25.The asterisk indicates the species known to occur first on the dunes and those following the expected trend for a pioneer species (Van Aarde *et al.* 1996b)

Community variables	Study site							
	10 year old rehabilitating site		14 year old rehabilitating site		18 year old rehabilitating site		Unmined forest of unknown age	
	Edge	Core	Edge	Core	Edge	Core	Edge	Core
Species richness	7.2 ± 0.3	5.0 ± 0.4	5.0 ± 0.3	5.2 ± 0.25	7.2 ± 0.4	10.0 ± 0.6	14.2 ± 0.5	13.7 ± 0.23
Shannon diversity	0.2 ± 0.01	0.2 ± 0.04	0.7 ± 0.02	0.9 ± 0.09	1.2 ± 0.05	1.5 ± 0.03	1.6 ± 0.05	1.5 ± 0.04
Number of individuals	46.5 ± 0.7	56.6 ± 9.1	5.8 ± 2.6	11.9 ± 0.5	16.0 ± 1.7	5.4 ± 0.4	23.1 ± 1.0	17.3 ± 1.7
Total number of species	8	8	5	5	12	10	14	15
Species composition	*Centrobolus fulgidus**	*Centrobolus fulgidus**	*Centrobolus fulgidus**	*Centrobolus fulgidus**	*Centrobolus fulgidus**	*Centrobolus fulgidus**	*Centrobolus fulgidus*	*Centrobolus fulgidus*
	*Centrobolus richardi**	*Centrobolus richardi**	*Centrobolus richardi**	*Centrobolus richardi**	*Centrobolus richardi**	*Centrobolus richardi**	*Centrobolus richardi*	*Centrobolus richardi*
	Doratogonus sp.	*Gnomeskelus tuberosus microdens*	*Gnomeskelus tuberosus microdens*	*Doratogonus sp.*	*Doratogonus* sp.	*Doratogonus* sp.	*Centrobolus rugulosus*	*Centrobolus rugulosus*
	Gnomeskelus tuberosus microdens	*Juliaformia* sp.1	*Orthoporoides* sp.	*Gnomeskelus tuberosus microdens*	*Gnomeskelus tuberosus microdens*	*Gnomeskelus tuberosus microdens*	*Doratogonus* sp.	*Doratogonus* sp.
	Juliaformia sp.1	*Juliaformia* sp. 2	*Spinotarsus* sp.*	*Spinotarsus* sp.*	*Juliaformia* sp.1	*Juliaformia* sp.1	*Gnomeskelus tuberosus microdens*	*Gnomeskelus tuberosus microdens*
	Juliaformia sp. 3	*Juliaformia* sp. 3			*Juliaformia* sp. 2	*Juliaformia* sp. 2	*Juliaformia* sp.1	*Juliaformia* sp.1
	Orthoporoides sp.	*Sphaerotherium giganteum*			*Juliaformia* sp. 3	*Juliaformia* sp. 3	*Juliaformia* sp. 2	*Juliaformia* sp. 2
	Spinotarsus sp.*	*Spinotarsus* sp.*			*Orthoporoides* sp.	*Sphaerotherium giganteum*	*Juliaformia* sp. 3	*Juliaformia* sp. 3
					Orthoporoides pyrocephalus	*Sphaerotherium* sp.	*Orthoporoides pyrocephalus*	*Orthoporoides pyrocephalus*
					Sphaerotherium giganteum	*Spinotarsus* sp.*	*Orthoporoides* sp.	*Orthoporoides* sp.
					Sphaerotherium punctulatum		*Sphaerotherium giganteum*	*Sphaerotherium giganteum*
					Spinotarsus sp.*		*Sphaerotherium punctulatum*	*Sphaerotherium punctulatum*
							Sphaerotherium sp.	*Sphaerotherium* sp.
							Spinotarsus sp.	*Spinotarsus* sp.
								Ulodesmus sp.

* It may be possible that *Juliaformia* sp. 1, 2 and 3 should be considered as one species.

Table 9 Mean (± standard error) edge width of *Centrobolus fulgidus* in the litter, shrub and tree strata of each study site, followed by the *F* and *p*-values.

Strata	Study site											
	10 year old rehabilitating site			14 year old rehabilitating site			18 year old rehabilitating site			Unmined forest of unknown age		
	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*
Litter	-	0.6	ns	19 ± 1.3	4.5	<0.1	34 ± 1.3	19.7	<0.01	10 ± 1.4	5.8	<0.1
Shrub	-	0.56	ns	19 ± 1.5	228.8	< 0.0001	18 ± 1.8	5.0	<0.02	20 ± 1.6	60.9	<0.01
Tree	-	0.55	ns	33 ± 1.7	13.4	<0.02	15 ± 1.3	4.2	<0.1	-	2.5	ns

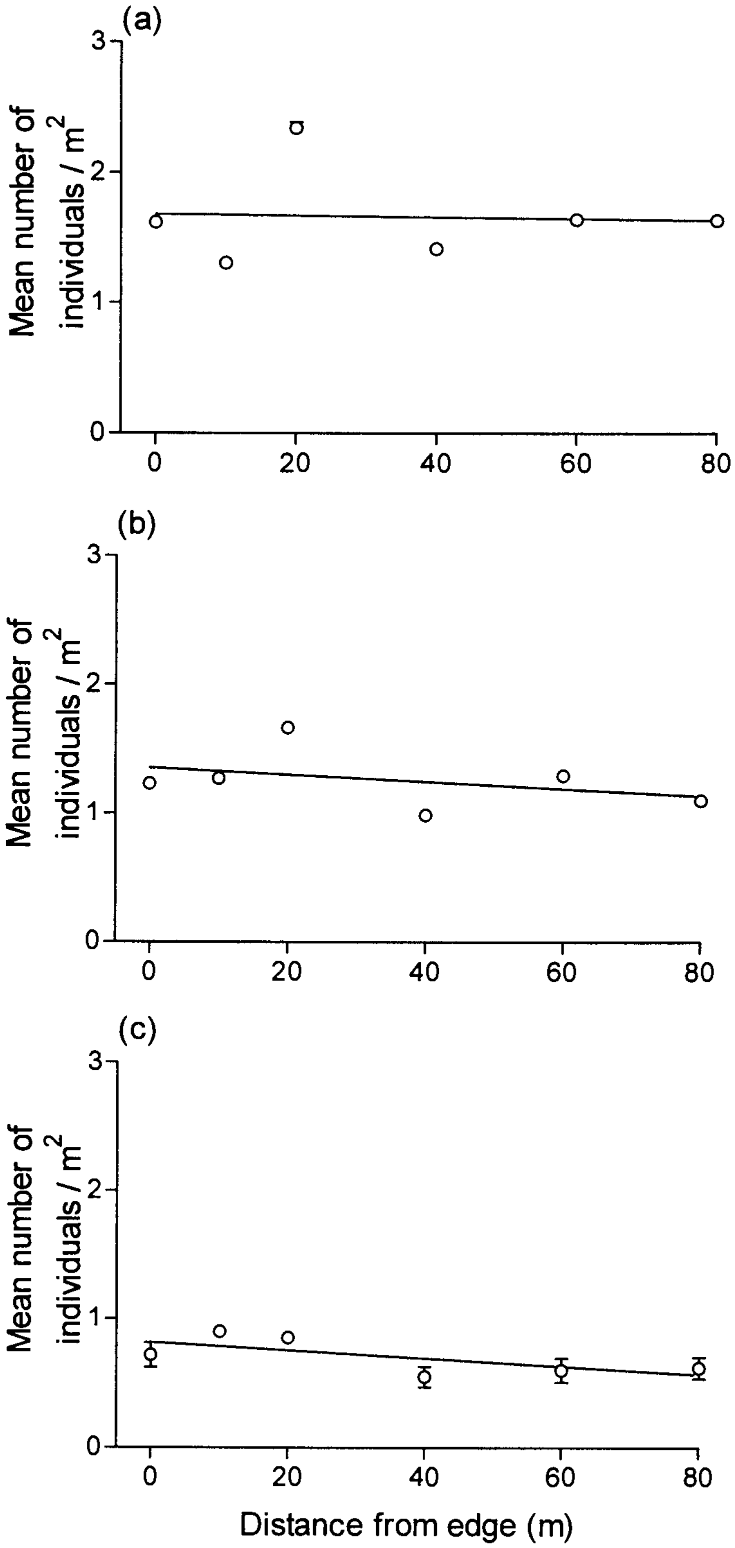

Figure 25 The influence of distance from edge on the mean (± standard error) number of individuals of *Centrobolus fulgidus* occurring on the (a) litter, (b) shrub and (c) tree strata in the 10 year old rehabilitating coastal dune forests. The dashed vertical lines indicate the calculated edge widths (see text p 16 and Table 9). Seperate linear regression lines were fitted for strata with significant edge widths. The regression equations are listed in Appendix 3.The standard errors were very low in the litter and shrub strata.

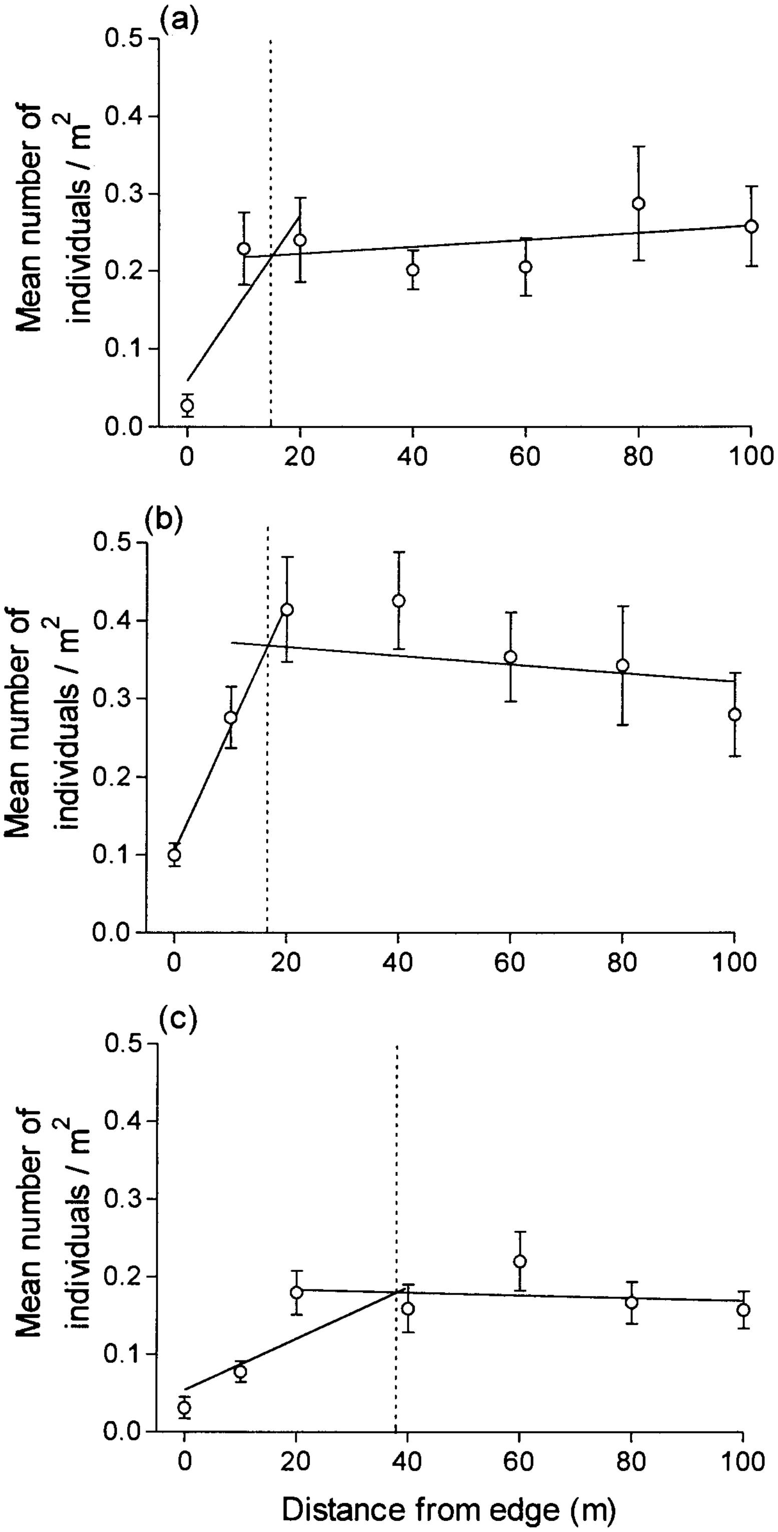

Figure 26 The influence of distance from edge on the mean (± standard error) number of individuals of *Centrobolus fulgidus* occurring on the (a) litter, (b) shrub and (c) tree strata in the 14 year old rehabilitating coastal dune forests. The dashed vertical lines indicate the calculated edge widths (see text p 16 and Table 9). Seperate linear regression lines were fitted for strata with significant edge widths. The regression equations are listed in Appendix 3.

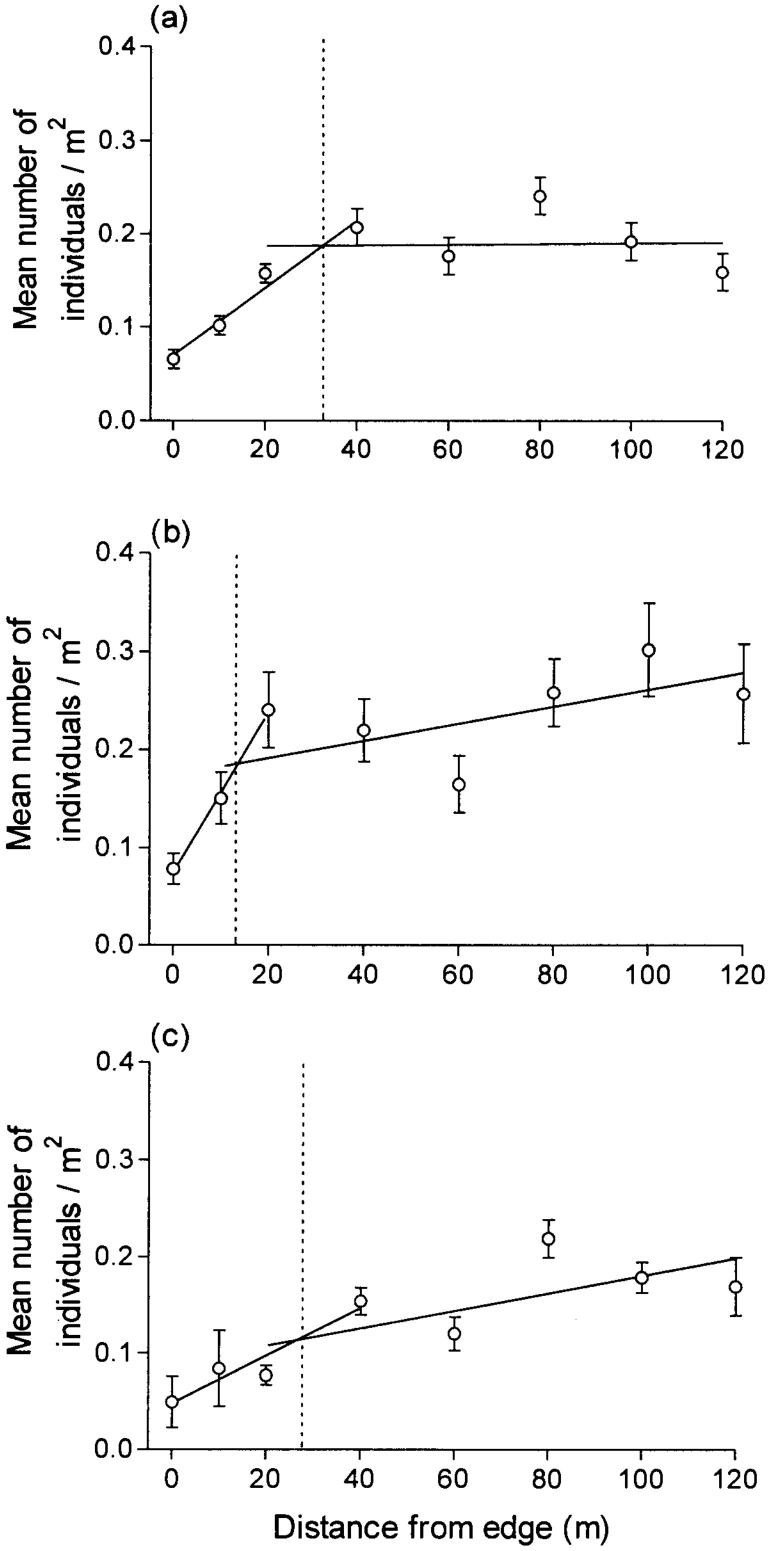

Figure 27 The influence of distance from edge on the mean (± standard error) number of individuals of *Centrobolus fulgidus* occurring on the (a)litter, (b) shrub and (c) tree strata in the 18 year old rehabilitating coastal dune forests. The dashed vertical lines indicate the calculated edge widths (see text p 16 and Table 9). Seperate linear regression lines were fitted for strata with significant edge widths. The regression equations are listed in Appendix 3.

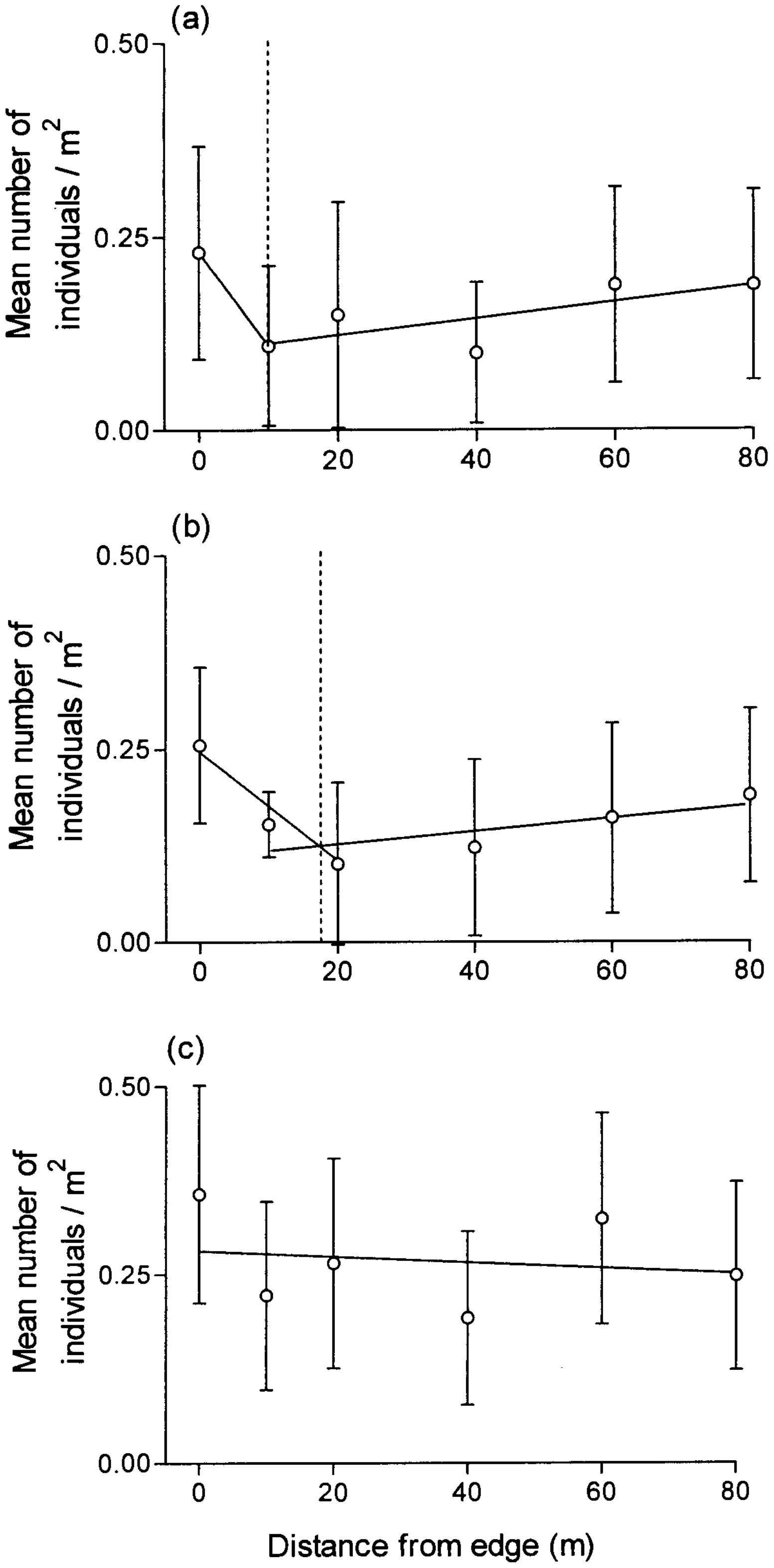

Figure 28 The influence of distance from edge on the mean (± standard error) number of individuals of *Centrobolus fulgidus* occurring in the (a) litter, (b) shrub and (c) tree strata in the unmined coastal dune forest. The dashed vertical lines indicate the calculated edge widths (see text p 16 and Table 9). Seperate linear regression lines were fitted for strata with significant edge widths. The regression equations are listed in Appendix 3.

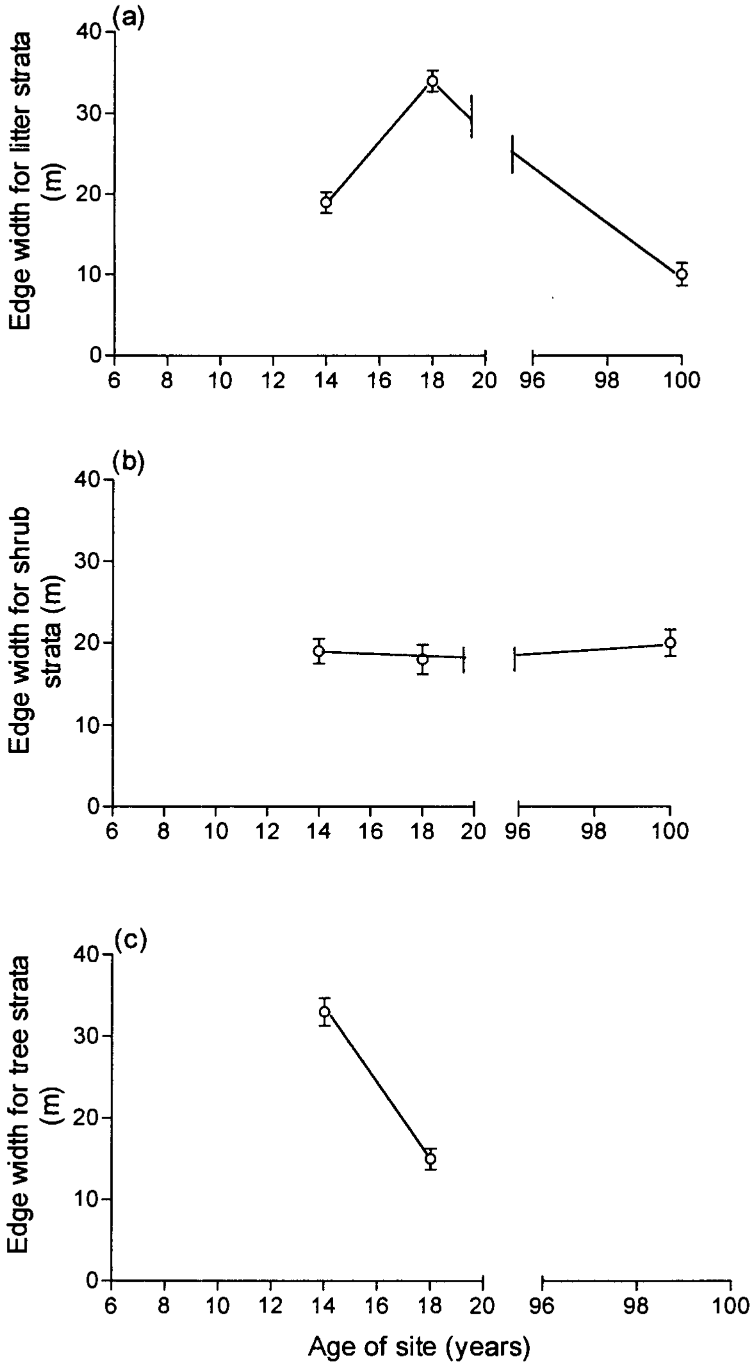

Figure 29 Mean (± standard error) calculated edge widths for the millipede strata (as described on p 16 and presented in Table 9) where (a) represents the litter strata, (b) the shrub strata and (c) the tree strata. In this figure the unmined forest is presented as 100 years old.

fragmenting the forests. The extent of the influence varies between the community variables, the community as a whole and regeneration age.

Species richness, diversity and number of individuals did not change significantly with increase in distance from the edge, but significant edge widths were seen in the 14 and 18 year old rehabilitating site as well as the unmined forest. One would expect a significant edge width considering millipede's dependency upon high moisture. Two years prior to the present study, MacCullum (*in prep.)* studied the influences of edge on temperature and humidity in the same regenerating sites as the present study (Chapter 2, pp 13 & 14). Significant edge effects were seen for humidity and temperature in the 10 and 14 year old rehabilitating sites, but not in the 18 year old site (Figures 3 & 4). The data of MacCullum (*in prep.*) was collected two years prior to the data collection of the present study, so the humidity and temperature trends may not be directly correlated to the current humidity and temperature trends, but it may in part explain the trends seen in the millipede community.

In the 10, 14 and 18 year old rehabilitating site edges are maintained by cutting vegetation along the sides of the roads, thereby allowing vehicles passage of roads. A constantly changing vegetation structure at the edge may seal the edge with more leaves and vegetation to transpire, while protecting the understory just inside the edge from desiccation (Camargo & Kapos 1995). The cutting of vegetation along the sides of the road as well as the influence of the continuing changes due to secondary succession causing gaps (ecological processes such as tree falls) may explain the absence of edge widths for the millipede community in the 18 year old rehabilitating site. The unmined forest consistently showed an edge width for all the community variables measured. The thick vegetation, especially at the edge, in the unmined forest may cause changes in microclimate variables at various distances, which may explain the significant edge width in the community variables.

In the results of the cluster analyses of the 10 year old regenerating site, 0 m was grouped with 40 m, 60 m and 80 m. *Doratogonus* sp. is alone responsible for 35% of the dissimilarity between the edge grid (0 m) and the grouping of 10 m with 20 m (Table 7). *Doratogonus* sp. also occurred at high densities in the rest of the distances grouped together in the core assemblage (40 m, 60 m and 80 m). The presence of this species therefore explains the grouping of 0 m with the core assemblage and not with the edge assemblage.

The distance of division into edge and core assemblages with multi - dimensional scaling techniques increased with increase in regeneration age (Figures 21 - 24), but the calculated edge widths did not show any trends with increase in regeneration age (Table 6).

Although the species composition differed between edge and core assemblages within the same study site, the number of species contributing to these assemblages were quite similar throughout all the study sites. In the 10 year old rehabilitating site, two species were identified as only occurring in the edge assemblage (*Doratogonus sp.* and *Orthoporoides* sp.), while two other species, (*Juliaformia* sp. 2 and *Sphaerotherium giganteum*) only occurred in the core assemblage. In the present study, there were also species occurring in the edge and core assemblages throughout all the regenerating and unmined sites, they were *Centrobolus fulgidus, Centrobolus richardi, Spinotarsus* sp. and *Gnomeskelus tuberosus microdens* (Table 8).

Furthermore, a few species were identified contributing only to the edge assemblage in the rehabilitating sites (*Orthoporoides* sp.). It is interesting to note that this species occurred in both assemblages in the unmined forest (Table 8). It could therefore be considered that the core assemblages of the regenerating sites have not developed sufficiently to meet the habitat requirements of this species.

In the unmined forest all the species contributing up to 50% of the cumulative dissimilarity between edge and core assemblages (Table 7) only contributed to the edge assemblage. The species were *C. fulgidus, Sphaerotherium punctulatum, Ulodesmus* sp., *G. tuberosus microdens* and *Orthoporoides* sp.. The difference in contribution of the same species to the edge or core assemblage between the study sites may be due to several reasons, which include change in interspecific interactions, the addition and /or replacement of species (Van Aarde *et al.* 1996b), changes in microclimate as well as the resources available. A change in resources available could influence the successional development causing changes in species assemblages.

C. fulgidus has been identified as a pioneer species due to a high density in the younger rehabilitating sites (Van Aarde *et al.* 1996b). Pioneer species will decrease and secondary species increase as the regenerating habitat ages (Van Aarde *et al.* 1996a). In an area with low food resources, pioneer species will pre - empt a large fraction of the total available resource, leaving only a small fraction for additional species (Dangerfield & Telford 1992). In the 10 year old rehabilitating site no significant edge widths were identified for *Centrobolus fulgidus*. This may be due to the interaction between *Doratogonus* sp. and *C. fulgidus* or due to the high density of *Doratogonus* sp. (Table 7). In the 14 and 18 year old regenerating and unmined study sites significant edge widths were identified for all three strata. In the 18 year old site, the edge width decreased from the litter strata to the tree strata. This may be ascribed to changes in microclimate between the different strata. The edge width in the shrub strata stayed relative constant with increase in regeneration age from 14 years. The shrub layer in the 14 year old regenerating site is relative prominent (see Table 1) and therefore the edge effect stayed constant with increase in regeneration age. In the tree strata no edge width could be identified in the 10 year old site and the unmined forest. The relative homogeneous vegetation in the 10 year old rehabilitating site may explain the absence of an edge width in this fragment. In the unmined forest, the high species richness of millipedes may cause the influence of the edge not to exert a significant edge effect on *C. fulgidus*. In the 14 and 18 year old rehabilitating sites the significant edge width seen may be due to less species being present, therefore *C. fulgidus* showed an edge width.

Although the results of the analyses to identify the edge influence is not consistent throughout the analyses, it can be concluded that the millipede community is influenced by roads acting as ecological edges. A definite edge and core assemblage identified confirms this statement. The edge widths differ between regenerating sites, but no specific trend was observed with increase in regeneration age.

Chapter 5

Edge effects in the rodent communities

Introduction

Small mammal community structures and densities may be influenced by roads (Adams & Geis 1983). Evidence from Brazilian Atlantic forest fragments suggested that species richness and diversity of rodents significantly increased with distance from the edge into the forest (Stevens & Husband 1998). However, disturbances at edges caused small mammal species richness in New South Wales to decrease with distance from the edge where the rate of decrease could be predicted by the extent of the disturbance (Dunstan & Fox 1996). In other studies, small mammal species were not affected by roads, suggesting that road surfaces were not inhibiting factors to their movements (Oxley *et al.* 1974), or that edges may even be used as travel routes (Bider 1968). No edge effects on small mammals were shown in highly fragmented landscapes in Southern Illinois (Heske 1995). All of these studies were conducted in different sites and included different species, therefore it could be expected that the final conclusions of reactions of small mammals towards edges would differ.

Rodent communities in the rehabilitating forests of northern KwaZulu - Natal are characterized by chronological changes in species composition, species richness, equitability and dominance due to differences in species - specific habitat preferences and interactions between species (Ferreira & Van Aarde 1997). The species composition between sites differed considerably, which resulted from the addition and/or replacement of species (Van Aarde *et al.* 1996a). Mean species diversity increased with age and attained a peak in the rehabilitating sites of 8 - 11 years of age, after which it declined (Ferreira 1993). Species richness increased with increasing age of rehabilitating site (Van Aarde *et al.* 1996a). *Mastomys natalensis* was identified as a pioneer species and apparently excludes other species from the young rehabilitated sites (1 - 7 years)

(Ferreira 1993). As a result of habitat changes, *Saccostomus campestris* co - occurs with *Mastomys natalensis* during the later stages of habitat regeneration (8 - 14 years) (Ferreira 1993). *Aethomys chrysophilus* commonly occurs in the older regenerating sites (Ferreira 1993). Ferreira & Van Aarde (1999) concluded that habitat variables can explain more of the variation in species - specific densities of a *Mastomys - Saccostomus - Aethomys* assemblage than species - interactions.

The present chapter is directed at assessing the influence of roads as edges on the community variables of rodent assemblages of different ages.

Materials and methods

Data collection

Rodents were trapped between 13 January and 5 March 1998 along the transect lines described for the bird surveys. All lines were sampled in a random order for five consecutive days and nights. One Sherman live-trap (7.5 X 9 X 23 cm) was set at 10 m intervals along each of the transect lines. All the traps were baited with oats, peanut butter and raisins and checked and reset at dawn. All individuals captured were identified to species level and marked (by toe-clipping) before being released. In the 10 year old site, the number of individuals caught was too low for sensible analyses and was thus excluded.

Data analyses

In order to estimate the distance specific relative densities of catches per trap for each study site, the frequencies of capture per trap for each distance in every cluster were calculated (Caughley 1977). The mean and standard errors for each distance were calculated for every cluster. The data describing the rodent community were analysed, edge effect quantified and community analysis done as described in Chapter 2 (p 16). Percentage contribution and consistency of contribution of different rodent species to between edge and forest core assemblages' dissimilarity and similarity could not be calculated due to no distinct assemblages identified through cluster analyses.

Population trends

The distance specific relative densities of catches per trap for each species were calculated using the same principles as for calculating distance specific relative densities of catches per trap for each study site, except that each species' data was analysed separately. The edge width was quantified as described in Chapter 2 (p 16). These analyses of population trends were used to investigate if any relationship existed between a specific species' density and distance from the edge in the different aged study sites.

Results

Community variables

Based on the slopes of least square linear regression lines used to estimate edge widths, species richness decreased significantly ($p < 0.05$) with increasing distance from the edge, both in the 14 and 18 year old site (Figure 30). In the unmined forest species richness decreased significantly ($p < 0.05$) with increase in distance from 98 m (Figure 31). Rodent species diversity decreased significantly (p <0.0.001) with increase in distance from edge in the 14 year old rehabilitating site, but showed no significant relationship with increasing distance from edge in the 18 year old rehabilitating and unmined forest (Figure 31). Rodent numbers recorded at sampling points decreased significantly ($p < 0.05$) with increase in distance from edge on the 14 and 18 year old rehabilitating and unmined sites (Figure 32).

Quantifying the edge width

A significant edge effect was identified for rodent species richness in the 14 year old rehabilitating ($F = 4.9$, $p < 0.02$, $e = 10$ m) and unmined site ($F = 2.3$, $p < 0.1$, $e = 98$ m). In the 18 year old regenerating site a significant edge was identified at 30 m ($F = 2.2$, $p < 0.1$) (Table 10). The edge width could not be interpreted in terms of regeneration age for species diversity and number of individuals due to the presence of only one significant edge width for each community variable (Table 10). The two significant edge widths for species richness were not shown in following aged sites (Table 10).

The community in space

Groups identified through cluster analyses had no clear relationship with distance from the road

Table 10 Mean (± standard error) edge width in metres (see text p 16) for species richness, species diversity and number of individuals for rodents in each study site, followed by the calculated *F* and *p* - values.

Community variable	Study site								
	14 year old rehabilitating site			18 year old rehabilitating site			Unmined forest of unknown age		
	Edge width	*F*	*p*	Edge width	*F*	*p*	Edge width	*F*	*p*
Species richness	10 ± 1.5	4.9	<0.02	-	1.3	ns	98 ± 1.3	2.33	<0.1
Species diversity	-	0.8	ns	30 ± 1.4	2.2	<0.1	-	0.9	ns
Number of individuals	-	15.3	ns	-	-1.7	ns	-	1.5	ns

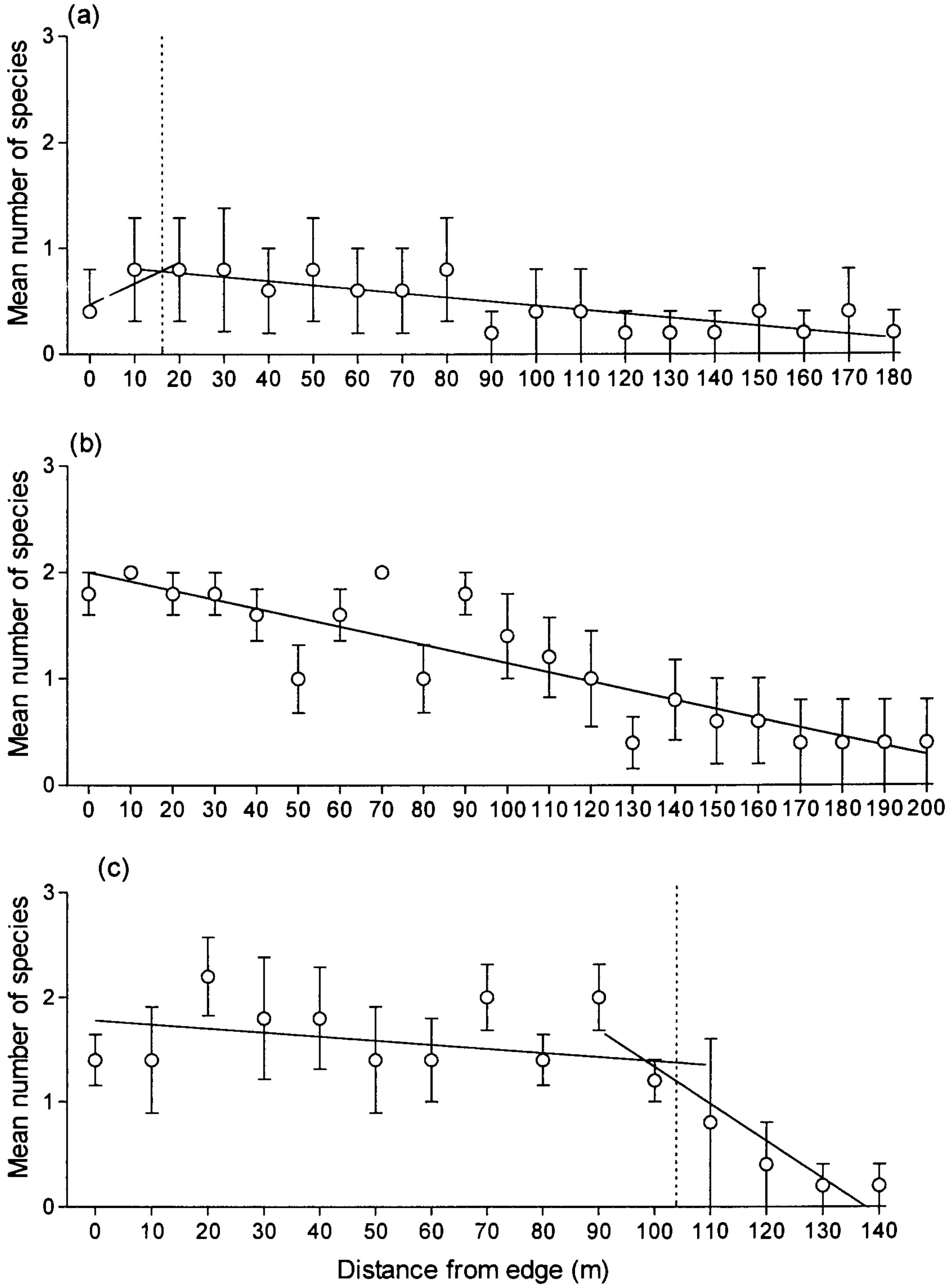

Figure 30 Mean (± standard error) species richness for rodents in regenerating and unmined coastal dune forests: the 14 year old site (a); the 18 year old site (b) and the unmined forest (c). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 10). The regression equations are listed in Appendix 5.

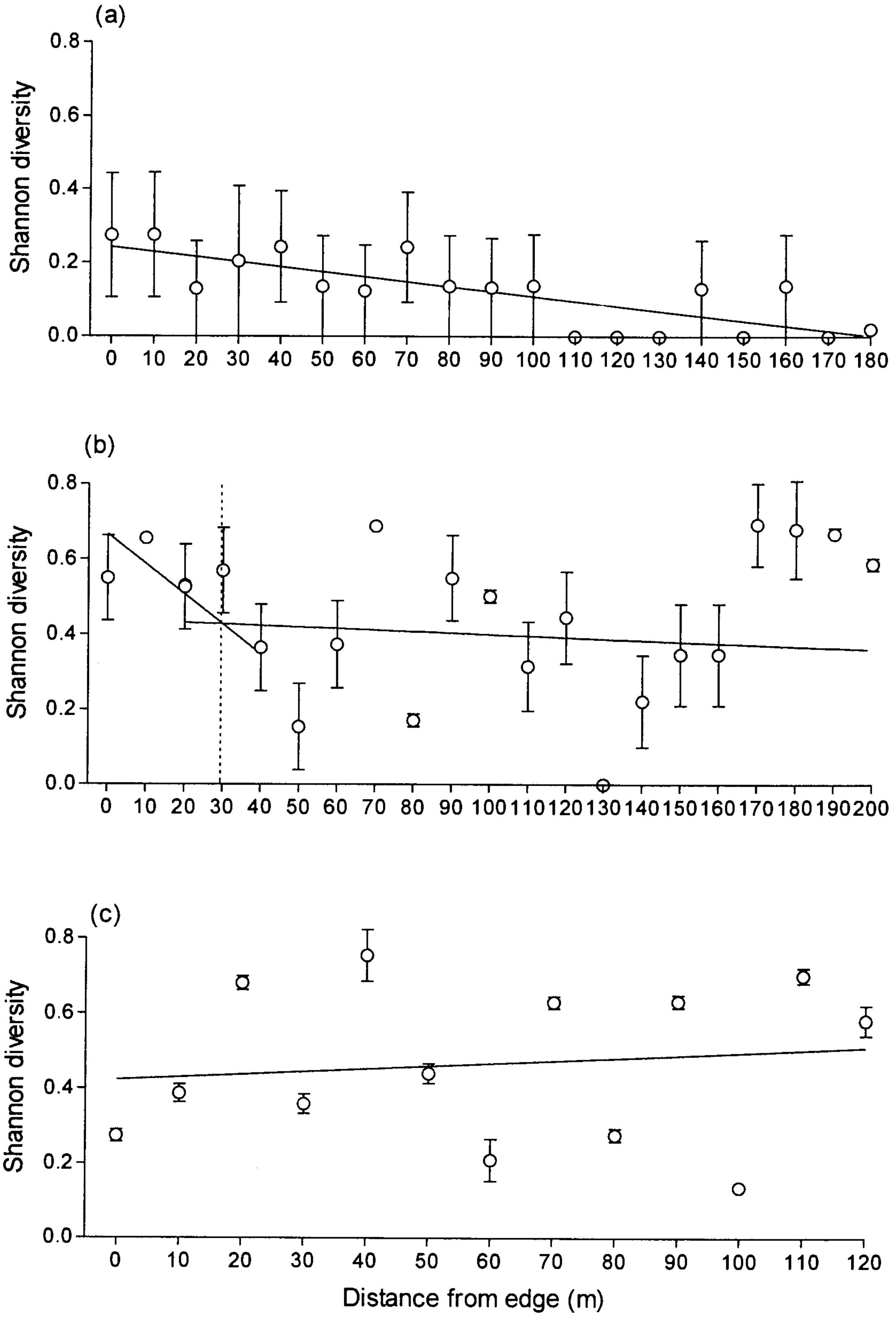

Figure 31 Mean (± standard error) species diversity for rodents in regenerating and unmined coastal dune forests: the 14 year old site (a); the 18 year old site (b) and the unmined forest (c). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 10). The regression equations are listed in Appendix 5.

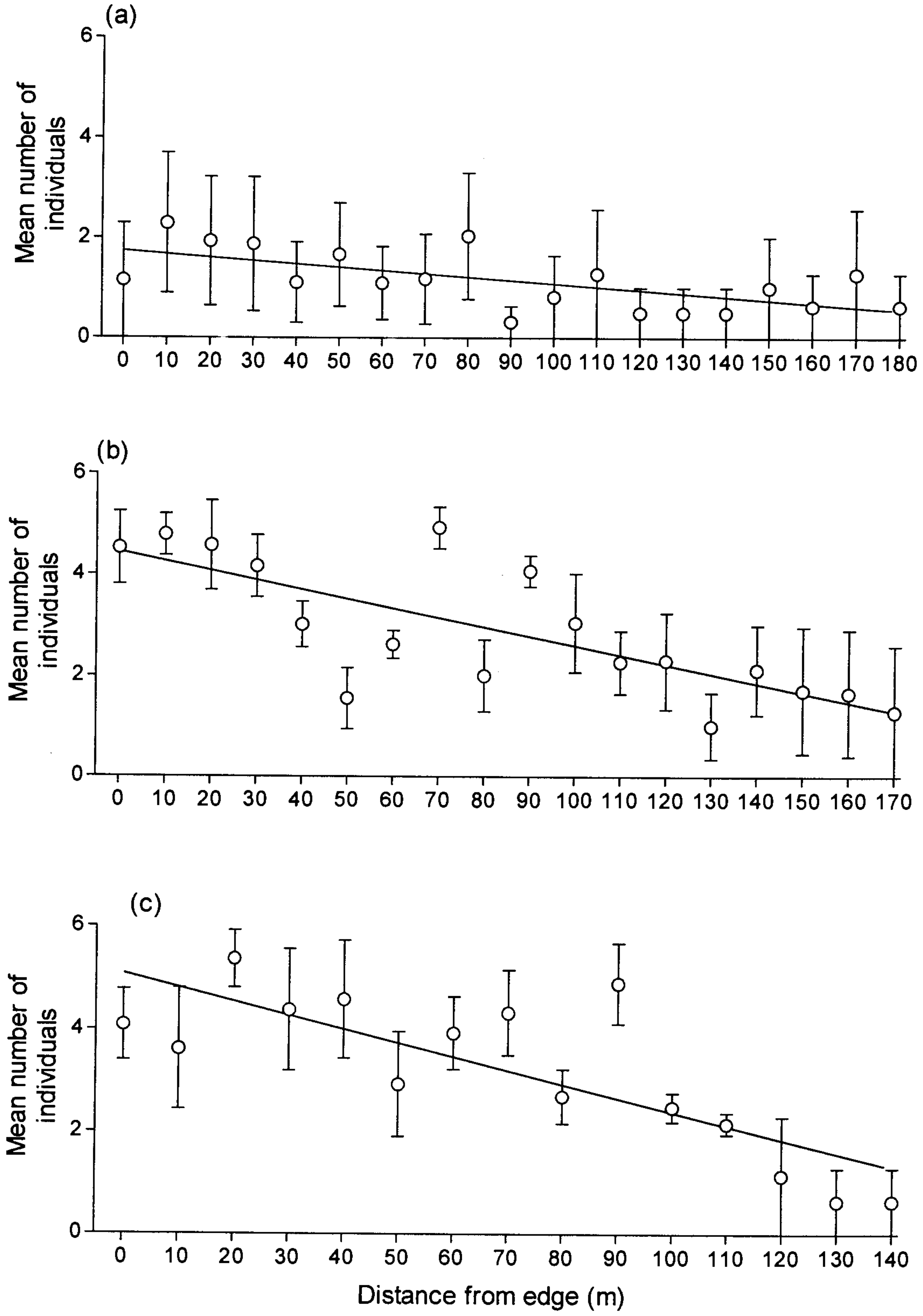

Figure 32 Mean (± standard error) number of individuals for rodents in regenerating and unmined coastal dune forests: the 14 year old site (a); the 18 year old site (b) and the unmined forest (c). The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for sites with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 10). The regression equations are listed in Appendix 5.

(Figures 33 - 35). Therefore no further community analyses was done.

Population trends

In figure 36, the two species with the highest density of catches per trap for each site are shown. In both the 18 year old site and unmined forest, *Aethomys chrysophilus* and *Saccostomus campestris* had the highest densities. In the 14 year old site, *S. campestris* and *Mastomys natalensis* had the highest density. In the 14 year old site, the density of catches per trap of *M natalensis* did not change significantly with increase in distance from edge, while densities of *S. campestris* changed significantly ($p < 0.1$) with increase in distance from edge (Figure 36). In the 18 year old regenerating site *S. campestris* and *A. chrysophilus* showed no significant trends with increase in distance from edge (Figure 36). In the unmined forest, *A. chrysophilus* numbers did not change significantly with increasing distance from edge, but *S. campestris* changed significantly ($p < 0.05$) with increase in distance from edge (Figure 36). In the 14 year old regenerating site, a significant edge width was identified for *S. campestris* at 34 m ($F = 8.78$, $p < 0.01$) (Table 11). In the 14 year old regenerating site, no significant edge width was identified for *M. natalensis*. In the 18 year old regenerating site, no significant edge width was identified for both species. In the unmined forest a significant edge width was identified for *S. campestris* ($F = 8.89$, $p < 0.01$, $e = 13$) (Table 11). Due to the absence of chronological following edge widths, no age related analysis was done with the species data.

A species list showing the distance specific relative mean density of catches per trap for each species in each rehabilitating and unmined site is provided in Appendix 6.

Discussion

Ferreira & Van Aarde (1997) conducted a study on the rodent communities in the same area as the present study and found that rodent density decreased significantly after 4 years of rehabilitation. In the present study, the sampled areas had lower densities of rodents compared to younger rehabilitating areas (A. C. Koekemoer *pers.comm.*). The younger sites in the same area (1 - 7 year old rehabilitating sites), were dominated by *M. natalensis*, while *S. campestris* dominated in the older sites (7 - 14 year old rehabilitating sites). In the unmined sites, *A. chrysophilus* dominated while *M. natalensis* and *S. campestris* occurred frequently (Ferreira &

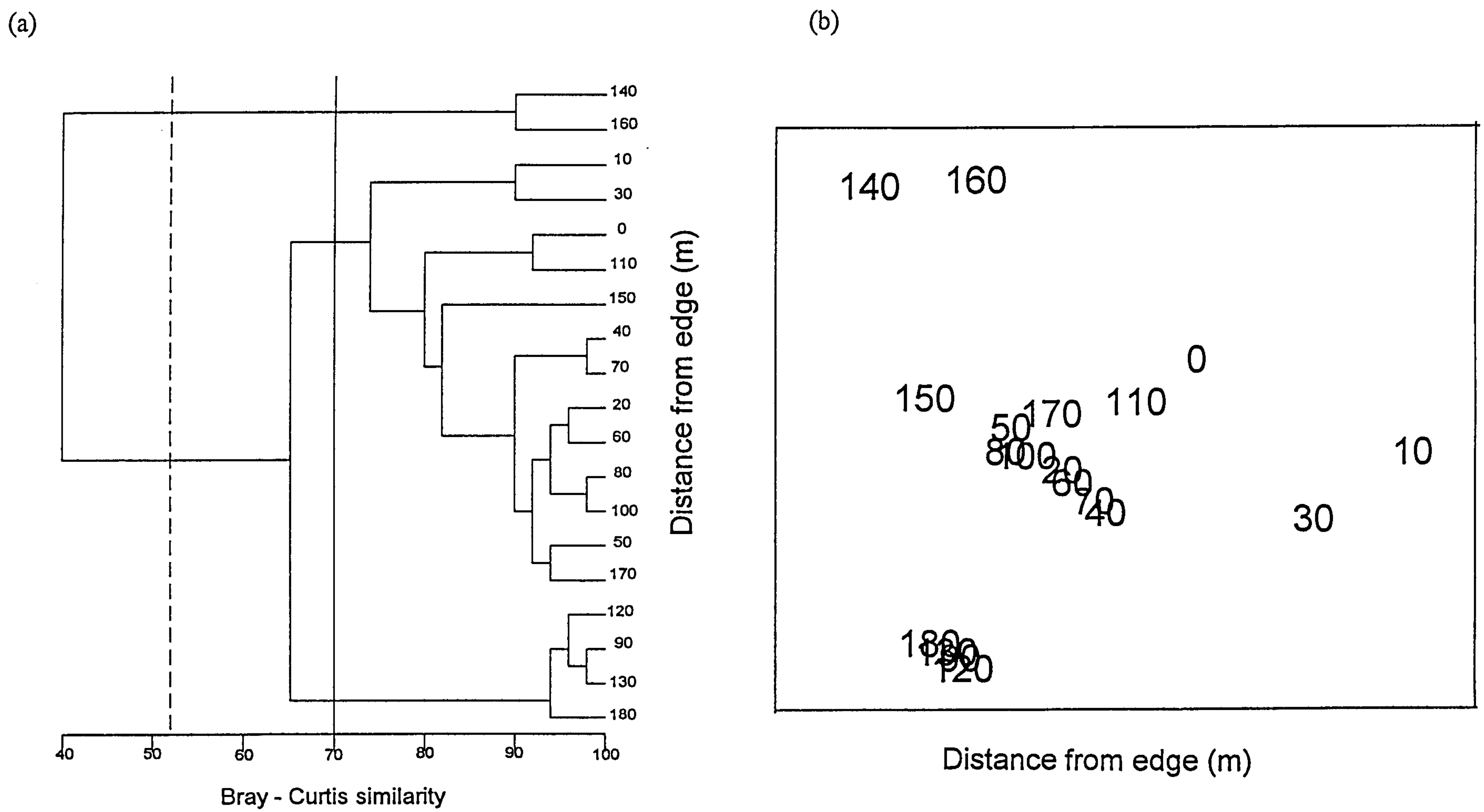

Figure 33 Dendrogram (a) and multi - dimensional scaling ordinations (MDS) in two - dimensions (b) of the distance related rodent data (fourth - root transformed) of the 14 year old site. Superimposed clusters from the dendrogram at similarity levels of 40% - 63% (dashed line) and 63% - 71% (continuous line) are indicated on the MDS (stress = 0.05). No groups were identified from the cluster analyses thus no groups are indicated on the MDS.

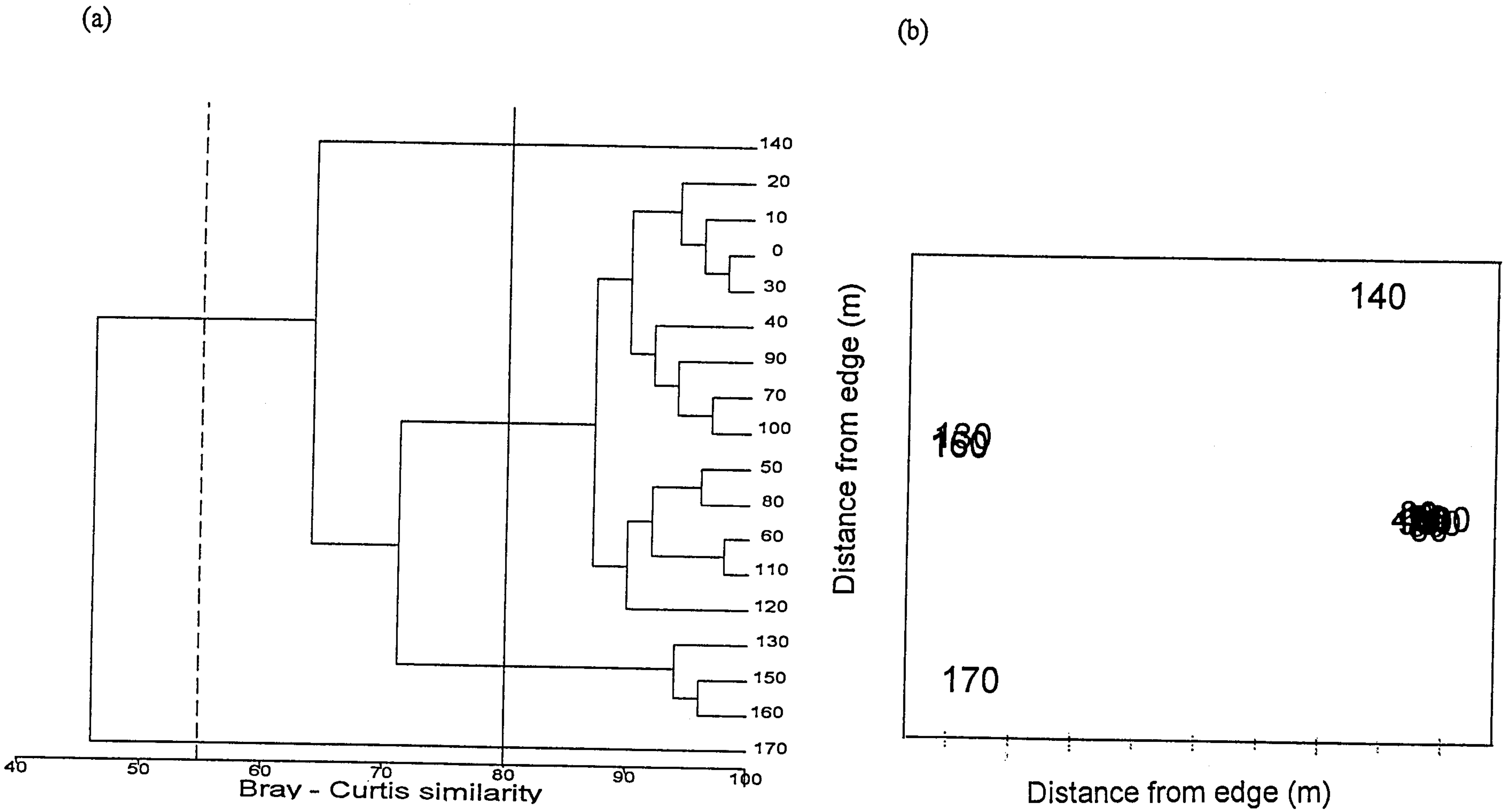

Figure 34 Dendrogram (a) and multi - dimensional scaling ordinations (MDS) in two - dimensions (b) of the distance related rodent data (fourth - root transformed) of the 18 year old site. Superimposed clusters from the dendrogram at similarity levels of 45% - 65% (dashed line) and 72% - 90% (continuous line) are indicated on the MDS (stress = 0.01). No groups were identified with the cluster analyses, thus no groups are indicated on the MDS.

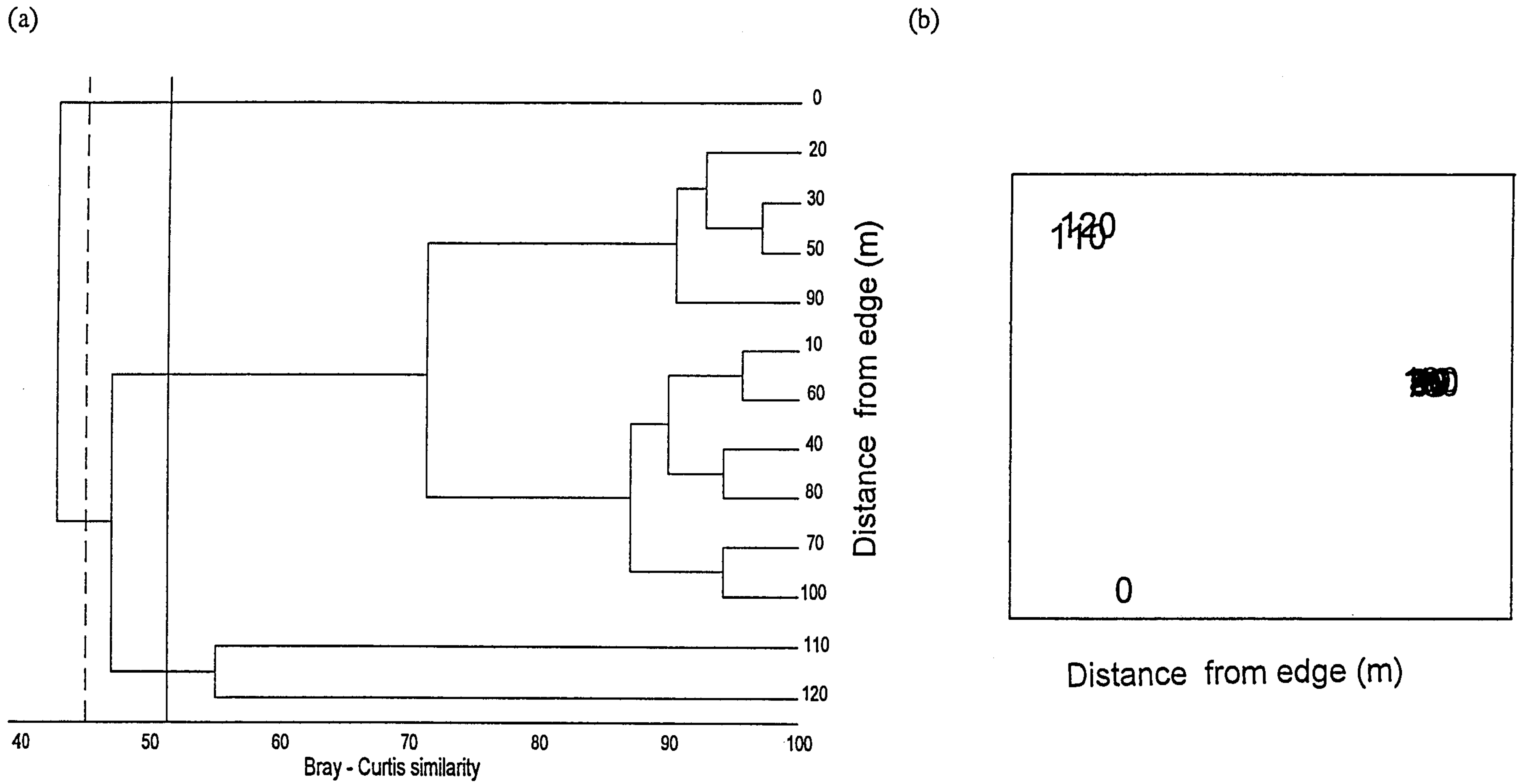

Figure 35 Dendrogram (a) and multi - dimensional scaling ordinations (MDS) in two - dimensions (b) of the distance related rodent data (fourth - root transformed) of the unmined forest. Superimposed clusters from the dendrogram at similarity levels of 43% - 47% (dashed line) and 47% - 55% (continuous line) are indicated on the MDS (stress = 0.01). No groups were identified from the cluster analyses thus no groups are indicated on the MDS.

Table 11 Mean (± standard error) edge widths in metres (see text p 16) for the two species in each study site with the highest relative densities of catches per trap, followed by the calculated *F* and *p* - values.

Species	Study site			Species	Study site					
	14 year old rehabilitating site				18 year old rehabilitating site			Unmined forest of unknown age		
	Edge width	*F*	*p*		Edge width	*F*	*p*	Edge width	*F*	*p*
Mastomys natalensis	-	1.9	ns	*Saccostomus campestris*	-	0.7	ns	13 ± 0.5	8.89	<0.01
Saccostomus campestris	34 ± 0.4	8.78	<0.01	*Aethomys chrysophilus*	-	-1.7	ns	-	1.5	ns

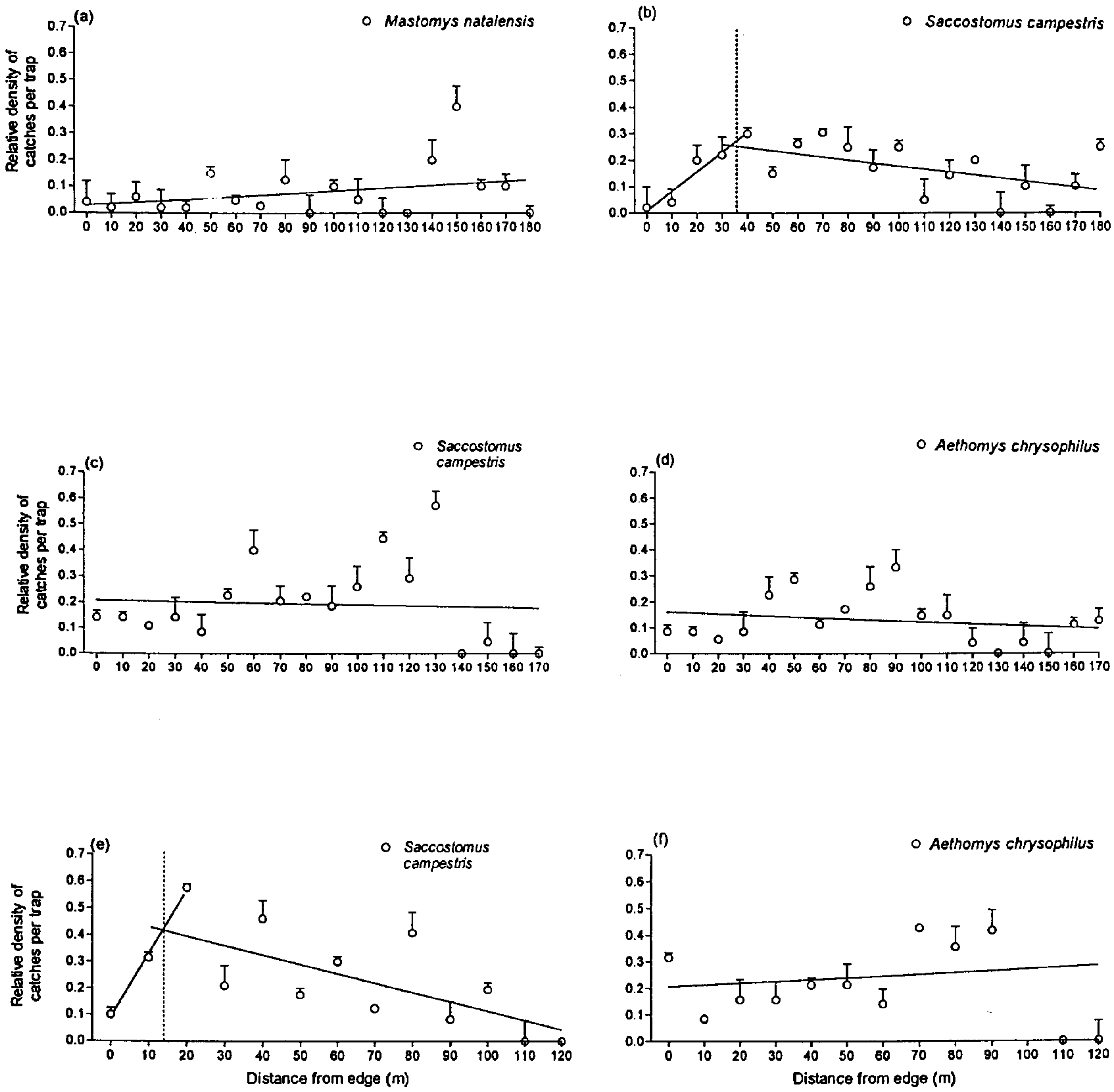

Figure 36 Mean (± standard error) relative densities of catches per trap in rehabilitating sites and unmined forest: *Mastomys natalensis* (a) and *Saccostomus campestris* (b) in the 14 year old rehabilitating site; *Saccostomus campestris* (c, e) and *Aethomys chrysophilus* (d, f) in the 18 year old rehabilitating and unmined sites, respectively. The regression lines were fitted and edge widths calculated as described on p 16. Seperate linear regression lines were fitted for species with significant edge widths. The dashed vertical lines indicate the calculated edge width (see Table 11). The regression line equations are listed in Appendix 5.

Van Aarde 1997). In the present study the same species dominated.

According to the results of the present study the rodent community was influenced by roads. It has been argued that edges function as a refugia for rodents during food scarcity (Hansson 1998) due to a higher diversity and richness of vegetation at edges in areas older than 20 years. In the present study a decrease in species richness and number of individuals occurred in sites with increasing distance from edges on areas younger than 20 years as well as edges older than 20 years. A decreasing species diversity with increase in distance from edge was only seen in the 14 year old rehabilitating site. Three significant edge widths were identified out of a possible nine throughout all the study sites and community variables. In Southern Illinois small mammals showed no edge effect in highly fragmented landscapes with distance from the edge, where the rate of decrease could be predicted by the extent of the disturbance (Dunstan & Fox 1996).

In the present study, no edge and core assemblages could be identified for the rodent community in any of the study sites using multi - variate analyses. For rodents, the current level of disturbance due to roads fragmenting the study site may indicate that there is no longer any true forest core. Other explanations of the absence of edge and core assemblages may be ascribed to vegetation throughout the study site not being sufficient to influence the population densities of rodents at edges (Heske 1995) and/or the influence of vegetation type on the other side of the edge (Swihart & Slyde 1984; Heske 1995; Stevens & Husband 1998).

The size of the roads may contribute to the variation in influence of edges on the distribution of small mammals (Oxley *et al.* 1974). Small mammals are reluctant to cross large, heavily - travelled roads (Oxley *et al.* 1974) as well as gravel roads (Meserve 1971; Kozel & Fleharty 1979), even as narrow as < 3 m (Swihart & Slade 1984). It has been shown that various small mammal species react differently to roads (Swihart & Slade 1984; Dunstan & Fox 1996; Laurence 1997). The natural histories of species depicts the home - ranges which influences the extent of the influence of edge effects. It has been shown that far - ranging species move relatively evenly over sites and adjoining areas, but species with limited movement ranges showed pronounced reactions to edges (Hansson 1998). Home-range sizes may influence the chance of a species moving across edges (Hansson 1998). Laurence (1997) showed a decrease in densities from the edge to forest interior of bush rats (*Rattus leucopus*).

Oxley *et al.* (1974) suggested that a clearance (e.g. road) would be important in inhibiting the movements of forest small mammals. Mammals adapted to open country would venture on to road surfaces, but such clearances would inhibit rodents to move into them. This may explain some of the trends of rodent species occurring in the rehabilitating and unmined forests of the present study. Although *Mastomys natalensis* is considered a pioneer species in regenerating forests (Ferreira 1993), it is known that this species has a wide habitat tolerance and generally does not occur in forests (Skinner & Smithers 1990). This could explain the absence of changes in relative density of catches per trap with increasing distance from edge in the 14 year old rehabilitating site (Figure 36).

The specific diet preferences of *Saccostomus campestris* may also explain the significant edge width for this species in the 14 year old rehabilitating site, causing the species to be indirectly influenced by edges. Although they eat grass and seed, they prefer larger seeds of forbs, bushes and trees (Skinner & Smithers 1990). But, in contrast *S. campestris* was not influenced by edges in the 18 year old regenerating site. The unmined forest showed a significant edge for *S. campestris* which may be linked to the species' specific diet preferences as mentioned previously. The absence of a significant change with increase in distance from edge, for *Aethomys chrysophilus* in the unmined as well as 18 year old regenerating site suggests that this species is not influenced by the edge in the more mature areas.

Although results from the analyses on the rodent community are in some conflict with each other, it can be concluded that the rodent community in the present study is influenced to some extent by roads. This can be concluded due to the decreasing trends in community variables with increase in distance from edge. It must be noted that these results are based on low density numbers of the rodent assemblage therefore distance related trends may be masked.

Chapter 6

Synthesis

Introduction

Edges are important factors affecting the recovery of species richness, species composition and species diversity of disturbed areas (Finegan 1996; Turner 1996; Turner & Corlett 1996). In spite of the numerous studies on edges (see Lovejoy *et al.* 1986; Bierregaard *et al.* 1992; Bolger *et al.* 1997), there has been little consensus on how to measure an edge, or the effects of edges on biotic and abiotic variables (Murcia 1995). As a consequence, the generality of the edge concept should be handled with caution when dealing with taxa comprising species with such diverse natural histories as in the present study (Heske 1995).

The present study aims at estimating the implications of roads passing through rehabilitating coastal dune forests at Richards Bay, for bird, millipede and rodent communities. It furthermore attempts to relate edge effects to successional development.

The two most important findings of the present study are that roads exert an edge effect on the bird, millipede and rodent communities, and that the effect of roads as edges change with successional age. Only the bird community was influenced by edge effects throughout all the analyses. The millipede and rodent communities did not show consistent trends of influence and extent throughout the analyses. These findings have implications for the rehabilitation program that gave rise to the development of the study area and also for other conservation programs.

Bird assembly variables for the rehabilitating sites decreased significantly with increasing distance from edge, but not in the unmined forest (Table 12). For birds significant edge widths were identified for all the sites except the 10 year old rehabilitating site. These edge widths suggests that for birds 14% to 38% of the regenerating sites were affected by roads (Table 13). Edge and core assemblages differed in species composition, with some species only contributing to the edge assemblages, while others only contributed to the core assemblages (see Table 4). Species richness and diversity of the edge assemblages decreased with an increase in regenerating age, but for the core assemblages increased with regenerating age. Thus with increasing regenerating age, the difference between edge and core assemblages decreased (Figure 37). In the study area, roads therefore caused an increase in species richness but a decrease in abundance in the younger rehabilitating sites and a lower species richness in the older sites (Figure 37).

Trends for the millipede community variables were not consistent throughout the analyses. Species richness, diversity and number of individuals did not change significantly with increasing distance from edge in the millipede community of all the study sites (Table 12). However, significant edge widths were identified for all the community variables of the 10 and 14 year old rehabilitating sites and unmined forest (Table 6). Identified edge and core assemblages indicated that roads do exert an edge effect on the millipede community throughout the study sites. In the 14 year old site 29% were considered as an edge area (Table 13). Nor species richness, diversity or number of individuals are responsible for distinctions into edge and core assemblages and the identification of edge and core assemblages were related to species composition. With millipede species richness and diversity increasing with regeneration age and the number of individuals decreasing (Figure 38), it can be concluded that a few species (pioneers) occur at high densities in the younger sites, while the other species occur in low densities. This pattern changes with an increase in regeneration age. Whth the number of species increasing as the densities of species decrease (see Figure 38) the trend is in agreement with that deviating from succession (Van Aarde *et al.* 1996b).

The variables of the rodent community generally decreased with increase in distance from edge (Table 12), but only three of the nine edge widths were significant. Up to 22% of the 18 year old rehabilitating site could be considered as edge (Table 13). However, no distinction could be made between edge and core assemblages (Table 14), probably due to the very low densities at which rodents were recorded during the present study.

The inability to calculate an edge width does not necessarily indicate the absence of an edge effect.

Table 12 Summary of overall trends of community variables with increasing distance from roads as edges. The trends are illustrated as negative "-" (variable decreases with an increase in the distance from edge) or no influence "0 " (variable showed no significant response to increasing distance from edge). None of the variables increased with distance from edge.

Age of study site	Taxon	Community variable		
		Species richness	Species diversity	Number of individuals
10 years old	Birds	-	-	-
	Millipedes	0	0	0
14 years old	Birds	-	-	-
	Millipedes	0	0	0
	Rodents	-	-	-
18 years old	Birds	-	-	-
	Millipedes	0	0	0
	Rodents	-	0	-
Unmined forest (unknown age)	Birds	0	0	0
	Millipedes	0	0	0
	Rodents	-	0	-

Table 13 Mean (± standard error) percentage area of the study sites influenced by edges for the bird, millipede and rodent communities based on the calculated edge widths (see page 16). Edge widths and size of study sites were used to estimate the percentage values.

Age of study site	Taxon	Community variable (% area influenced by edges)		
		Species richness	Species diversity	Number of individuals
10 years old	Birds	-	-	-
	Millipedes	18 ± 0.06	-	-
14 years old	Birds	35 ± 0.08	-	38 ± 0.02
	Millipedes	-	17 ± 0.09	29 ± 0.08
	Rodents	11 ± 0.1	-	-
18 years old	Birds	21 ± 0.1	19 ± 0.07	14 ± 0.04
	Millipedes	-	-	-
	Rodents	-	22 ± 0.05	-

Table 14 Summary of results of multivariate techniques to distinguish between an edge and core assemblage based on data of bird, milliped and rodent communities.

Age of study site	Taxon	Edge and core assemblages identified?
10 years old	Birds	yes
	Millipedes	yes
14 years old	Birds	yes
	Millipedes	yes
	Rodents	no
18 years old	Birds	yes
	Millipedes	yes
	Rodents	no
Unmined forest (unknown age)	Birds	yes
	Millipedes	yes
	Rodents	no

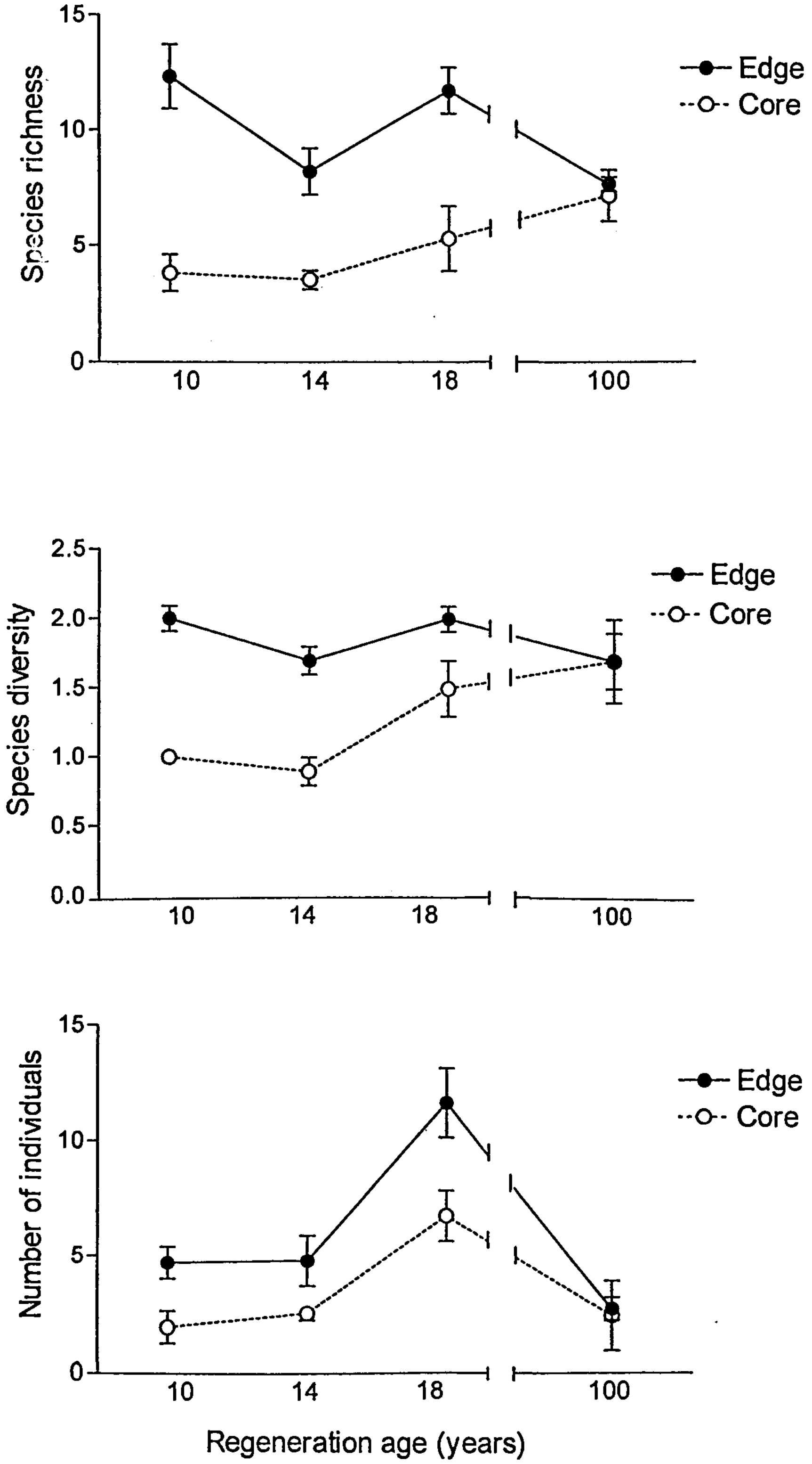

Figure 37 Edge widths for (a) species richness, (b) species diversity and (c) number of individuals of the bird edge and core assemblages in relation to regeneration age. The unmined forest is presented as 100 years old.

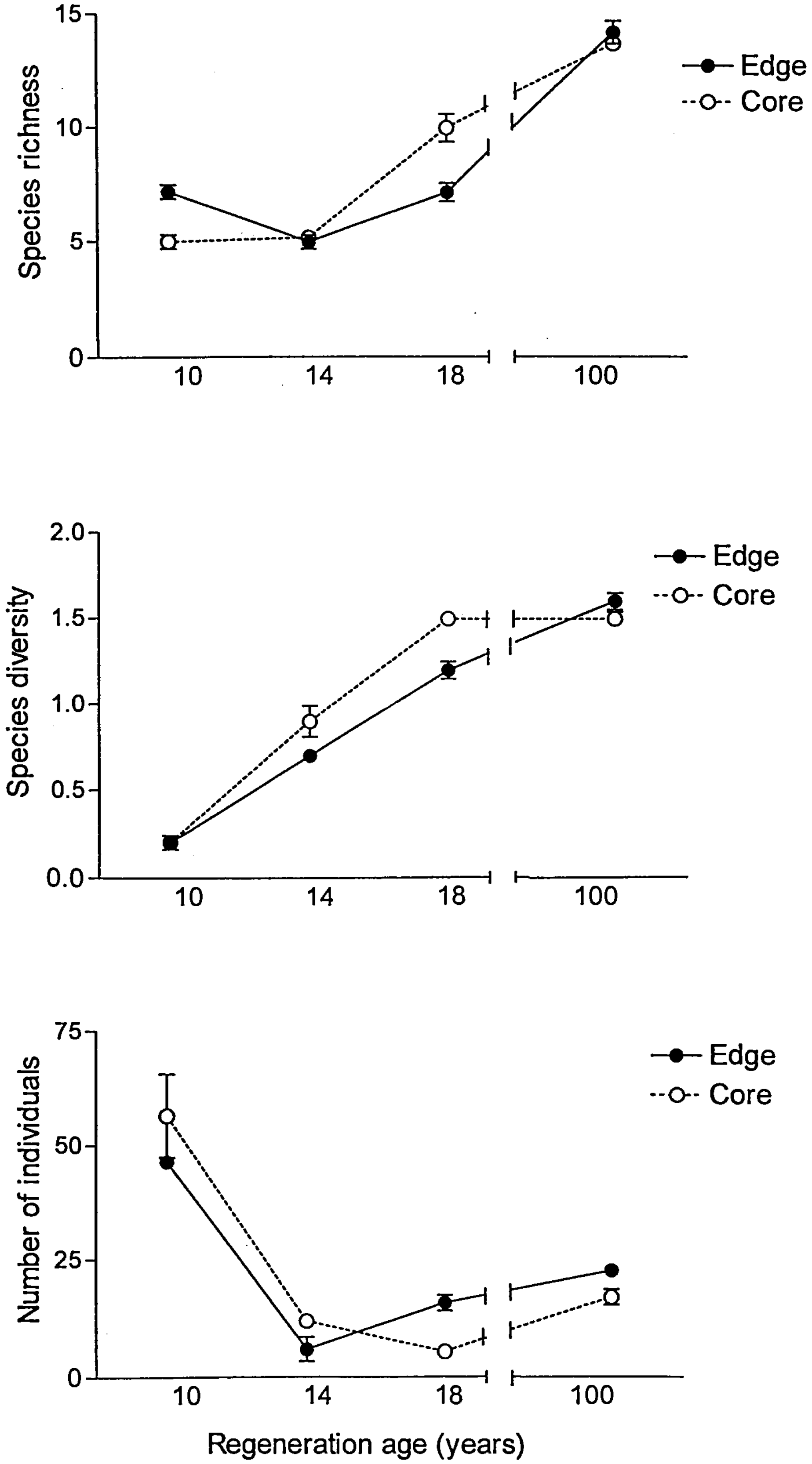

Figure 38 Edge widths for (a) species richness, (b) species diversity and (c) number of individuals of the millipede edge and core assemblages in relation to regeneration age. The unmined forest is presented as 100 years old.

I am therefore inclined to give more importance to the discriminating ability of multi - variate analyses, and the significance of community variable trends with increasing distance from the edge. In general, the results from the present study indicate that roads as edges influence the bird, millipede and rodent communities.

Several influences may mask distance related edge effects. Studies like the present, should possibly be conducted every few years in order to consider annual differences in environmental factors. Though roads as edges may affect faunal assemblages, it cannot be considered as a disturbance factor affecting successional pathways. The species of edge assemblages include the pioneer species (see Tables 4 and 8), but these pioneer species may also occur in the core assemblages. However, not all species colonizing edges are core species. High species heterogeneity could result in a higher colonization rate of rehabilitating areas.

Based on the results of the present study, roads may give rise to changes on the local level. In some cases the local assemblages have unique species, but in other it is merely a change in abundance of the constituent species. Roads contribute to the local heterogeneity. In coastal dune forests, such heterogeneity may alter species composition and relative abundances of species. Although a specific trend with increasing regeneration age was not always clear, definite differences between the sampling sites were apparent. The unmined forest may be considered as consisting of a mosaic of successional stages, and is thus influenced by a combination of edge effects. These edges may therefore explain some or most of the existing differences in community characteristics between rehabilitating and unmined coastal dune forests.

The present study could not provide a generalisation of the results across taxa probably due to the different roles being played by the taxa in the ecosystem, which includes factors such as the habitat requirements and specific diet preferences of species within each taxon.

Summary

Since 1977 coastal dunes north - east of Richards Bay (28°43′S, 32°12′E) have been mined by Richards Bay Minerals for heavy metals such as zircon and ilmenite (Van Aarde *et al.* 1996a). The mining process results in the total destruction of all dune vegetation. Post mining rehabilitation has resulted in the development of a known age sere of coastal dune forests. These rehabilitating forests are fragmented by a network of tracks and roads. The present study deals with the influence of these roads (considered as edges) on the regenerating bird, millipede and rodent communities and potential changes in such influences with increasing successional age.

The study is based on surveys conducted in rehabilitating forests of 10, 14 and 18 years of age and an unmined forest of unknown age. Data on bird and rodents was collected from using five cluster groups, of five transect lines each, in every study site. Transect lines within a cluster group were 10 m apart and the cluster groups within a site were at least 50 m apart. The lengths of the transect lines varied between 80 m and 210 m due to the variation of width in the sites. At each point, bird observations lasted 12 minutes. The first two minutes were used to allow the birds to become accustomed to my presence, while the last 10 minutes were spent on actual observations. To establish the distance of the bird from the observer, an imaginary line was drawn from the bird to the ground and the distance estimated using a calibrated "Ranging" range finder (Telinject Ranging TLR 75).

Data on the millipedes were collected as described by Van Aarde *et al.* (1996b). Quadrats (3 X 15 m) were set out in transect lines at the edge (0 m), 10 m, 20 m, 40 m, 60 m, 80 m, 100 m or 120 m depending on site width. There were 13, 12, 19 and 14 transect lines sampled in the 10, 14, 18 year old rehabilitating and unmined forest respectively.

Rodents were trapped along the transect lines described for bird sampling between 13 January and 5 March 1998. One Sherman - live trap (7.5 X 9 X 23 cm) was set at 10 m intervals along each of the transect lines for five consecutive days and nights. The traps were baited with oats, peanut butter and raisins and checked and reset at dawn. All individuals captured were identified and

marked before being released. In the 10 year old site, the number of individuals were too low for sensible analyses and was therefore excluded.

The community variables used were species richness (number of species), Shannon Wiener diversity index and number of individuals. The edge width was defined as the distance to the point where the apparent influence of the road (edge) on the variable changed. A p - value obtained from a technique to quantify the edge width (adapted from a two - phase regression model of Nickerson, *et. al.*1989) was used to indicate if a significant edge width existed or not. The data were divided into two groups using all possible pairs, e. g. 0 m - 10 m and 20 m - 120 m or 0 m - 20 m and 30 m - 120 m and error sums of squares calculated for all these possible pairs of linear regressions, with no overlap between pairs. For each of these data sets a separate linear regression with a value on the abscissa of the division point was calculated. The division point with i) the lowest collective error sum of squares and (ii) a significant regression for at least one group, was accepted as the best estimate of the edge width (Matlack 1993; Fox *et al.* 1997).

Multi - variate analyses, (cluster analyses and multi - dimensional scaling) were used to distinguish between edge and core assemblages. All data were fourth - root transformed to ensure the most sensible biological interpretation of the multi - dimensional scaling and cluster analysis results (Clarke & Warwick 1994). If edge and core assemblages were identified, further analyses were done to identify species contributing to these assemblages.

Distance from edge did affect species richness, diversity and number of individuals in the regenerating coastal dune bird communities due to the decreasing trends with increase in distance from edge. Significant edge widths were identified for species richness, species diversity and number of individuals in the 18 year old and unmined forest. No significant edge widths were identified in the 10 year old rehabilitating sites, and in the 14 year old rehabilitating site significant edge widths were only identified for species richness and number of individuals. The edge width estimated for richness and number of individuals decreased with increase in regeneration age (from 14 years and older).

Bird species, such as Blackeyed Bulbul, Tambourine Dove and Collared Sunbird, that only occur in edge assemblages, are representative of typical mixed habitats. Species contributing to both edge and core assemblages included the Bleating Warbler and Olive Sunbird, which are typical of bird parties and forage in the middle and upper strata. Species such as the Sombre Bulbul only contributed to the edge assemblage of the unmined forest, reflecting on the differences between rehabilitating and unmined sites. The number of nectarivores and insectivores decreased with increase in distance from the edge, and significant edge widths existed for all feeding guilds in the 14 and 18 year old rehabilitating sites. It is therefore concluded that bird communities in these coastal dune forests are influenced by the roads fragmenting the sites.

The millipede community showed no variables decreasing or increasing trends with increasing distance from roads. Significant edge widths were identified for millipede species richness in the 10 year old rehabilitating site for species diversity and number of individuals in the 14 year old rehabilitating site. These results may be ascribed to the influence of the vegetation being cleared at the road edges. The cutting of vegetation to allow vehicles the usage of roads opens the forest canopy conceivably and may influence humidity and temperature (see Figures 3 & 4). In the 18 year old rehabilitating site no significant edge widths were identified, which may be due to the combination of clearing of roads and gaps in the canopy created by tree falls. These influences may enhance variability and hide potential edge effects. Significant edge widths were identified for all three community variables in the unmined forest.

The distinct edge and core assemblages in the millipede community, identified through multi - variate analyses may be ascribed to different contributions of species to these assemblages. Some species only occurred in the edge assemblages of the 10 year old site (e.g. *Doratogonus* sp. and *Orthoporoides* sp.) with other species only contributing to the core assemblage of the same site (e. g. *Juliaformia* sp. 1 and *Sphaerotherium giganteum*). Differences in species composition of the edge and core assemblages could also be seen in other study sites. Significant edge widths were identified for *Centrobolus fulgidus* in the litter, shrub and tree strata of the 14 and 18 year old rehabilitating and unmined sites. In the 10 year old rehabilitating site, no significant edge widths were identified, and in the unmined forest significant edge widths were identified for the litter and shrub strata.

General decreasing trends in rodent richness, diversity and number of individuals were seen with increasing distance from the edge in all the study sites, but significant edge widths could only be estimated for three of nine possible community variables. No edge and core assemblages could be identified by using multi - variate analyses. In the 14 year old site *Saccostomus campestris* showed a significant edge width. In the 18 year old site, no species showed a significant edge width, and in the unmined forest, a significant edge width could only be identified for *S. campestris*.

Based on the results of the present study, both the null hypotheses, (distance does not affect the species richness and diversity of the regenerating coastal dune bird, millipede and rodent assemblages, and age of regenerating forest does not influence the edge effect), are rejected. It can therefore be concluded that roads (considered as edges) exert an effect on the bird, millipede and rodent communities, and that there are differences between the extent of the influence in the different aged sites for the three taxa studied. These results have implications for this specific rehabilitation program as well as other conservation projects.

It is therefore clear that edges could explain some of the dissimilarities in community characteristics between regenerating and unmined forest communities. Although roads considered as edges may affect faunal assemblages, it cannot be considered as a disturbance factor affecting the successional pathways. For this specific rehabilitation program, pioneer species occurring in the edge assemblages may also occur in the core assemblages. The edge assemblages reflect on an high of heterogeneity and could therefore contribute to the ecological view of local heterogeneity. In the case of coastal dune forests, such heterogeneity alter the species composition and relative abundances of species. It may thus cause regenerating forests to deviate in characteristics from the relatively undisturbed forests.

Opsomming

Sedert 1977 word kusduine noord - oos van Richards Baai (28°43'S, 32°12'E) deur Richards Baai Minerale gemyn vir swaar metale. Hierdie mynproses vereis die totale vernietiging van alle duin plantegroei. Rehabilitasie na die mynproses, veroorsaak 'n reeks van ontwikkelende kusduin woude met bekende ouderdomme. Hierdie rehabiliterende kusduin woude word gefragmenteer deur 'n netwerk van paaie. Die huidige studie behandel die invloed van hierdie paaie (beskou as randte) op die rehabiliterende voël-, duisendpoot- en knaagdiergemeenskappe en die potentiële veranderinge met toename in suksessionele ouderdom.

Die huidige studie is gebaseer op opnames uitgevoer in rehabiliterende woude van 10, 14 and 18 jaar oud en 'n ongemynde bos van onbekende ouderdom. Voël- en knaagdiermonsterneming was gedoen deur die gebruik van vyf groepe bestaande uit vyf transeklyne elk. Transeklyne in 'n groep was 10 m apart en groepe in 'n ouderdomstand was ten minste 50 m apart. Die lengtes van die transeklyne het gewissel van 80 m tot 210 m as gevolg van die verskille in breedtes van die studie areas. Die monsterneming was gedoen in 'n ewekansige volgorde. Waarnemings het bestaan uit 12 minute by elke afstands punt, waarvan die laaste 10 minute gespandeer was aan voëlopnames. Die eerste twee minute was gebruik om die voëls te laat gewoond raak aan die teenwoordigheid van die waarnemer. Om die afstand vanaf die voël na die waarnemer te bepaal, is 'n denkbeeldige lyn getrek vanaf die voël na die grond, en die afstand is bepaal deur gebruik te maak van 'n "Ranging range finder" (Telinject Ranging TLR 75).

Die duisendpootmonsterneming was gedoen soos beskryf deur Van Aarde *et al.* (1996a). Kwadrante (3 X 15 m) was uitgesit in transeklyne by die rand (0 m), 10 m, 20 m, 40 m, 60 m, 80 m, 100 m en/of 120 m afhangende van die ouderdomstand se breedte. Daar was onderskeidelik 13, 12, 19 en 14 transeklyne gemonster in die 10, 14 en 18 jaar oue rehabiliterende en ongemynde bosse.

Een Sherman val (7.5 X 9 X 23 cm) was gestel op 10 m intervalle op elke transeklyn vir vyf opeenvolgende dae en nagte. Die valle was gestel met lokaas bestaande uit 'n kombinasie van hawermout, grondboontjiebotter en rosyntjies. Al die individue wat gevang was, is geïdentifiseer en gemerk voordat hulle weer vrygelaat is. In die 10 jaar oue stand was daar te min individue gevang om die data sinvol te interpreteer, dus was die stand se muisdata nie ingesluit in data analises nie.

Spesies rykheid (aantal spesies), Shannon Wiener diversiteit en die hoeveelheid individue teenwoordig was gebruik as gemeenskapsveranderlikes. Die randbreedte was gedefinieër as die afstand tot by die punt waar die pad (rand) 'n invloed het op die spesifieke gemeenskapsveranderlike. Deur gebruik te maak van die aangepaste dubbel - fase regressie model van Nickerson *et al.* (1989), is die betekenisvolheid van die rand breedte bepaal. Die data was verdeel in twee groepe deur gebruik te maak van alle afstandspare, bv. 0 m - 10 m en 20 m - 120 m, of 0 m - 20 m en 30 m - 120 m en die fout som van kwadrante was bereken vir elke moontlike paar liniêre regressies sonder enige oorvleueling van die pare. Vir elkeen van hierdie data groepe was 'n aparte liniêre regressie bereken wat die waarde van die snypunt op die verdelingspunt bereken het. Die verdelingspunt met (I) die kleinste gesamentlike fout som van kwadrate, en (ii) 'n betekenisvolle regressie vir ten minste elke groep, was aanvaar as die beste skatting van die breedte van die randeffek (Matlack 1993; Fox *et al.* 1997).

Meerveranderlike analises was gebruik om indien moontlik, natuurlike rand en kern groeperings uit te ken. Alle data was vierde mags wortel getransformeer voor enige analises uitgevoer is, om te verseker dat die mees sinvolle biologiese interpretasie gemaak kan word vanaf die resultate van die meerveranderlike analises (Clarke & Warwick 1994). Indien 'n rand en kern groepering geïdentifiseer is, is verdere analises gedoen om die spesies wat bygedrae het tot die verdeling te identifiseer.

Die voëlgemeenskap toon afnemende neigings in die gemeenskaps veranderlikes met toename in afstand vanaf die rand. Dit toon aan dat afstand vanaf die rand die spesies rykheid, diversiteit en aantal individue beinvloed. Betekenisvolle randwydtes is vir spesies rykheid, diversiteit en aantal individue in die 18 jaar oue en ongemynde bos geïdentifiseer. Daarenteen is geen betekenisvolle randwydtes in die 10 jaar oue rehabiliterende stand geïdentifiseer nie. In die 14 jaar oue stand was daar slegs betekenisvolle randwydtes geïdentifiseer vir spesies rykheid en aantal individue. Die randwydte geïdentifiseer vir spesies rykheid en aantal individue neem af met toename in ouderdom van stand (slegs vanaf die 14 jaar oue stand).

Voëlspesies soos Swartoogtiptol, Witborsduifie en Kortbeksuikerbekkie in die rand groepering is verteenwoordigend van spesies wat in gemendge habitatte voorkom. Spesies wat bydraes gemaak het tot beide die rand en kern groeperings, het ingesluit die Kwê - kwêvoël en Olyfsuikerbekkie, wat kenmerkend is van voëlsamekomste en wat gewoonlik in die middel en boonste strata voed. Spesies soos die Gewone Willie het slegs bygedrae tot die rand groepering van die ongemynde bos, en dit reflekteer op die verskille tussen die rehabiliterende stande en ongemynde bos. Die aantal insektivore en nektarivore het afgeneem met toename vanaf die rand, en betekenisvolle randwydtes is bereken vir al die voedingsgildes in die 14 en 18 jaar oue rehabiliterende stande. Hieruit word afgelei dat voëlgemeenskappe in die kusduinwoude beinvloed word deur paaie wat die areas fragmenteer.

Die duisendpootgemeenskap in die rehabiliterende kusduin woud word beinvloed deur paaie. Die snoei van plantegroei op die randte verseker voertuie die gebruik van die bospaaie. Dit mag die toename in humiditeit en temperatuur veroorsaak (Figure 3 & 4). Betekenisvolle randwydtes is geidentifiseer vir spesies rykheid in die 10 jaar oue rehabiliterende stand en vir spesies diversitet en aantal individue in die 14 jaar oue rehabiliterende stand. Hierdie resultate kan ook toegskryf word aan die snoei van plantegroei op die randte. In die 18 jaar oue rehabiliterende stand was daar geen betekenisvolle randwydtes geïdentifiseer nie. Dit kan moontlik wees as gevolg van die kombinasie van die snoei van die rand plantegroei en die gapings in die blaredak as gevolg van bome wat omval. Hierdie gebeure verhoog die veranderlikheid en verbloem die potensiële rand effekte. Betekenisvolle randwydtes was geïdentifiseer vir al drie gemeenskapsveranderlikes in die ongemynde bos.

In die duisendpootgemeenskap word daar onderskeid getref tussen definitiewe rand- en kerngroeperings deur meerveranderlike analises. Sommige spesies kom slegs in die rand groepering van die 10 jaar oue rehabiliterende stand voor (bv. *Doratogonus* sp. en *Orthoporoides* sp.), terwyl ander spesies slegs bydrae tot die kerngroepering van die betrokke ouderdomstand (bv. *Juliaformia* sp. 1 en *Sphaertotherium giganteum*). Hierdie verskille in spesiesamestelling van die rand- en kerngroeperings kan ook in die ander ouderdomstande gesien word. Betekenisvolle randwydtes is in die kompos-, struik en boom strata in die 14 en 18 jaar oue rehabiliterende en ongemynde stande geïdentifiseer. In die 10 jaar oue rehabiliterende stand is geen betekensivolle randwydtes geïdentifiseer nie, maar wel in die ongemynde bos vir kompos- en struikstrata.

Algemene afnemende neigings in die knaagdier rykheid, diversiteit en aantal individue was in al die studie areas waargeneem met toename in afstand vanaf die rand. Betekenisvolle randwydtes was slegs vasgestel vir drie gemeenskapsveranderlikes in die drie ouderdomstande. Geen rand of kern groeperings kon geïdentifiseer word met die meerveranderlike analises nie. In die 14 jaar oue stand kon slegs *Saccostomys campestris* 'n betekenisvolle randwydte toon. In die 18 jaar oue stand het die dominante spesie nie 'n betekenisvolle randwydte getoon nie, en in die ongemynde bos is daar slegs 'n betekenisvolle randwydte vir *S. campestris* geïdentifiseer.

Gebaseer op die resultate van die huidige studie, kan beide die nul hipoteses (eerstens, dat afstand vanaf die rand nie die spesies rykheid, diversiteit en aantal individue van die voël-, duisendpoot- en knaagdiergemeenskap beinvloed nie, en tweedens dat die ouderdom van rehabiliterende stande nie die randeffek beinvloed nie) verwerp word. Die feit dat paaie (beskou as randte) 'n invloed uitoefen op die voël-, duisendpoot- en knaagdiergemeenskappe, en dat daar verskille is tussen die mate van invloed op die verskillende ouderdomstande word beskou as die twee belangrikste gevolgtrekkings. Hierdie resultate het implikasies vir die spesifieke rehabilitasie program asook vir ander bewaringsprojekte.

Dit word dus duidelik dat randte sommige van die verskille in gemeenskapskenmerke tussen regenererende en ongemynde bos gemeenskappe kan verduidelik. Dit blyk dat alhoewel paaie wat as randte beskou word, die fauna gemeenskappe beinvloed, kan dit nie beskou word as 'n faktor wat die suksessionele paaie beinvloed nie. Vir hierdie spesifieke rehabiliteringsprogram kan pionierspesies in die rand sowel as kerngemeenskap voorkom. Die randgemeenskappe reflekteer 'n hoë vlak van heterogeniteit en kan dus bydrae tot die ekologiese beskouing van lokale heterogeniteit. In die geval van kusduinwoude sal sulke heterogeniteit die spesiessamestelling en relatiewe digthede van spesies beinvloed. Dit kan dus veroorsaak dat rehabiliterende en relatiewe onversteurde bosse verskil.

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Appendix 1 Regression equations of the variables and feeding guilds for the bird community of each study site. Separate linear regressions were fitted for sites (or feeding guilds) with significant edge widths.

Community variables	Study site			
	10 year old rehabilitating site	14 year old rehabilitating site	18 year old rehabilitating site	Unmined forest of unknown age
Species richness	$y = 2.6 - 0.01x$ (Df = 9; $p < 0.0001$)	$y = 11.8 - 0.1x$ (Df = 16; $p < 0.1$)	$y = 13.8 + 0.03x$ (Df = 14; $p < 0.02$)	$y = 14 - 0.7x$ (Df = 9; $p < 0.02$)
		$y = 7.5 - 0.03x$ (Df = 16; $p < 0.0001$)	$y = 16.4 - 0.07x$ (Df = 14; $p < 0.0001$)	$y = 4 + 0.04x$ (Df = 9; ns)
Species diversity	$y = 2.7 - 0.01x$ (Df = 9; $p < 0.0001$)	$y = 1.9 - 0.008x$ (Df = 16; $p < 0.0001$)	$y = 2.1 + 0.006x$ (Df = 14; ns)	$y = 3 - 0.09x$ (Df = 9; $p < 0.001$)
			$y = 2.4 - 0.006x$ (Df = 14; $p < 0.01$)	$y = 1.7 + 0.0003x$ (Df = 9; ns)
Number of individuals	$y = 2.6 - 0.09x$ (Df = 9; $p < 0.01$)	$y = 8.8 - 0.2x$ (Df = 16; $p < 0.0001$)	$y = 12.8 + 0.2x$ (Df = 14; $p < 0.02$)	$y = 4.5 - 0.2x$ (Df = 9; $p < 0.01$)
		$y = 2.5 - 0.0004x$ (Df = 16; ns)	$y = 16.4 - 0.07x$ (Df = 14; $p < 0.001$)	$y = 2.4 - 0.004x$ (Df = 9; ns)
Feeding guilds				
Insectivores	$y = 0.1 - 0.003x$ (Df = 9; ns)	$y = 0.2 - 0.04x$ (Df = 16; $p < 0.05$)	$y = 0.2 + 0.003x$ (Df = 14; $p < 0.05$)	$y = 0.07 - 0.0002x$ (Df = 9; ns)
	$y = 0.07 - 0.0002x$ (Df = 9; $p < 0.02$)	$y = 0.008 - 0.0003x$ (Df = 16; $p < 0.05$)	$y = 0.3 - 0.001x$ (Df = 14; $p < 0.001$)	
Nectarivores	$y = 0.1 - 0.0007x$ (Df = 9; $p < 0.02$)	$y = 0.2 - 0.003x$ (Df = 16; $p < 0.02$)	$y = 0.2 + 0.005x$ (Df = 14; ns)	$y = 0.05 + 0.00003x$ (Df = 9; ns)
		$y = 0.07 - 0.0002x$ (Df = 16; $p < 0.02$)	$y = 0.3 - 0.001x$ (Df = 14; $p < 0.001$)	
Frugivores	$y = 0.1 - 0.001x$ (Df = 9; $p < 0.02$)	$y = 0.2 - 0.004x$ (Df = 16; $p < 0.02$)	$y = 0.2 + 0.005x$ (Df = 14; ns)	$y = 0.06 - 0.0001x$ (Df = 9; ns)
		$y = 0.1x - 0.0001x$ (Df = 16; $p < 0.02$)	$y = 0.2 - 0.0009x$ (Df = 14; $p < 0.001$)	

* Df indicates the degrees of freedom

Appendix 2 Species specific bird abundances obtained through observations on transect lines in the study sites. These are the mean (± standard error) values for the edge and core assemblages for each species.

Order and family	Common name	Scientific name	Date of description	Age of study site							
				10 years		14 years		18 years		Unknown age (unmined forest)	
				Edge	Core	Edge	Core	Edge	Core	Edge	Core
Order Falconiformes											
Family Accipitridae											
	Cuckoo Hawk	*Aviceda cuculoides*	Swainson 1837	0.00	0.00	0.00	0.00	0.20 ± 0.20	0.00	0.00	0.00
	Long crested Eagle	*Lophaetus occipitalis*	Daudin 1800	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02 ± 0.02
	Lizard Buzzard	*Kaupifalco monogrammicus*	Temminck 1824	0.00	0.00	0.00	0.00	0.02 ± 0.02	0.00	0.00	0.00
	African Goshawk	*Accipiter tachiro*	Latam 1790	0.00	0.00	0.02 ± 0.02	0.00	0.00	0.00	0.00	0.00
Order Columbiformes											
Family Columbidae											
	Redeyed Dove	*Streptopelia semitorquata*	Rüppell 1837	0.00	0.00	0.02 ± 0.02	0.00	0.00	0.00	0.00	0.00
	Tambourine Dove	*Turtur tympanistria*	Temminck & Knip 1810	0.01± 0.01	0.00	0.03 ± 0.03	0.07 ± 0.07	0.07 ± 0.04	0.00	0.07 ± 0.07	0.00
Order Musophagiformes											
Family Musophagidae											
	Livingstone Lourie	*Tauraco livingstonii*	Gray 1864	0.00	0.00	0.02 ± 0.02	0.00	0.00	0.00	0.00	0.02 ± 0.02
Order Cuculiformes											
Family Cuculidae											
	European Cuckoo	*Cuculus canorus*	Linnaeus 1758	0.01 ± 0.01	0.00	0.00	0.00	0.00	0.00	0.10 ± 0.07	0.00
Order Strigiformes											
Family Strigidae											
	African Scops Owl	*Otus senegalensis*	Smith 1834	0.00	0.00	0.00	0.00	0.03 ± 0.03	0.00	0.00	0.00
Order Alcediniformes											
Family Alcedinidae											
	Pygmy Kingfisher	*Ispidina picta*	Boddaert 1783	0.01 ± 0.01	0.00	0.00	0.07 ± 0.07	0.00	0.00	0.00	0.00
Family Meropidae											
	Little Bee-eater	*Merops pusillus*	Müller 1776	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03 ± 0.03
Order Coraciiformes											
Family Bucerotidae											
	Crowned Hornbill	*Tockus alboterminatus*	Büttikorfer 1889	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02 ± 0.02
Order Piciformes											
Family Lybiidae											
	White-eared Barbet	*Stactolaema leucotis*	Sundevall 1850	0.00	0.00	0.00	0.00	0.00	0.00	0.13 ± 0.13	0.06 ± 0.06
	Goldenrumped Tinker Barbet	*Pogoniulus bilineatus*	Sundevall 1850	0.03 ± 0.03	0.13 ± 0.03	0.03 ± 0.03	0.00	0.17 ± 0.06	0.23 ± 0.20	0.00	A0.04 ± 0.04

Appendix 2 continued on next page...

Appendix 2 (continued)

Order and family	Common name	Scientific name	Date of description	Age of study site							
				10 years		14 years		18 years		Unknown age (unmined forest)	
				Edge	Core	Edge	Core	Edge	Core	Edge	Core
Family Picidae											
	Black Cuckooshrike	*Campephaga flava*	Vieillot 1817	0.00	0.00	0.00	0.00	0.03 ± 0.03	0.17 ± 0.17	0.00	0.00
Family Dicruridae											
	Forktailed Drongo	*Dicrurus adsimilis*	Bechstein 1794	0.03 ± 0.03	0.03 ± 0.03	0.00	0.01 ± 0.01	0.60 ± 0.26	0.07 ± 0.07	0.00	0.06 ± 0.04
	Squaretailed Drongo	*Dicrurus ludwigii*	Smith 1834	0.10 ± 0.06	0.00	0.16 ± 0.06	0.03 ± 0.02	0.25 ± 0.15	0.13 ± 0.08	0.00	0.18 ± 0.11
Family Pycnonotidae											
	Blackeyed Bulbul	*Pycnonotus barbatus*	Desfontaines 1787	0.03 ± 0.03	0.00	0.03 ± 0.03	0.07 ± 0.07	0.10 ± 0.07	0.00	0.13 ± 0.06	0.47 ± 0.21
	Terrestrial Bulbul	*Phyllastrephus flavostriatus*	Swainson 1837	0.00	0.00	0.00	0.00	0.04 ± 0.03	0.00	0.00	0.00
	Sombre Bulbul	*Andropadus importunus*	Vieillot 1818	0.08 ± 0.04	0.00	0.00	0.05 ± 0.04	0.05 ± 0.05	0.00	0.33 ± 0.07	0.32 ± 0.08
	Yellowbellied Bulbul	*Chlorocichla flaviventris*	Smith 1834	0.07 ± 0.03	0.00	0.03 ± 0.03	0.14 ± 0.11	0.03 ± 0.03	0.20 ± 0.2	0.00	0.02 ± 0.02
	Yellowspotted Nicator	*Nicator gularis*	Hartlaub & Finsch 1870	0.01 ± 0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Family Turdidae											
	Natal Robin	*Cossypha natalensis*	Smith 1840	0.07 ± 0.03	0.00	0.07 ± 0.04	0.00	0.42 ± 0.30	0.13 ± 0.13	0.13 ± 0.13	0.04 ± 0.03
	Whitebrowed Robin	*Erythropygia leucophrys*	Vieillot 1836	0.18 ± 0.12	0.02 ± 0.02	0.00	0.00	0.00	0.00	0.00	0.02 ± 0.02
Family Sylviidae											
	Yellowbreasted Apalis	*Apalis flavida*	Strickland 1852	0.66 ± 0.13	0.28 ± 0.10	0.82 ± 0.30	0.45 ± 0.10	1.76 ± 0.3	1.45 ± 0.60	0.33 ± 0.18	0.11 ± 0.05
	Rudd's Apalis	*Apalis ruddi*	Shelley 1908	0.03 ± 0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Longbilled Crombec	*Sylvietta rufescens*	Vieillot 1817	0.02 ± 0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Rattling Cisticola	*Cisticola chiniana*	Smith 1843	0.00	0.00	0.07 ± 0.07	0.00	0.00	0.00	0.00	0.00
	Bleating Warbler	*Camaroptera brachyura*	Vieillot 1820	1.23 ± 0.20	0.63 ± 0.36	1.67 ± 0.40	1.20 ± 0.30	3.27 ± 0.37	1.77 ± 0.59	0.47 ± 0.17	0.36 ± 0.11
	Tawnyflanked Prinia	*Prinia subflava*	Gmelin 1789	0.05 ± 0.02	0.00	0.10 ± 0.10	0.01 ± 0.01	0.43 ± 0.13	0.17 ± 0.17	0.00	0.00
Family Muscicapidae											
	Spotted Flycatcher	*Muscicapa striata*	Pallas 1764	0.00	0.00	0.00	0.00	0.12 ± 0.06	0.07 ± 0.07	0.00	0.00
	Dusky Flycatcher	*Muscicapa adusta*	Boie 1828	0.00	0.00	0.17 ± 0.11	0.00	0.18 ± 0.05	0.00	0.07 ± 0.07	0.00
	Blue-grey Flycatcher	*Muscicapa caerulescens*	Hartlaub 1865	0.45 ± 0.14	0.00	0.20 ± 0.20	0.08 ± 0.08	0.20 ± 0.04	0.2 ± 0.21	0.00	0.00
	Fantailed Flycatcher	*Myioparus plumbeus*	Hartlaub 1858	0.02 ± 0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Whitetailed Flycatcher	*Trochocercus albonotatus*	Sharpe 1891	0.00	0.00	0.00	0.00	0.03 ± 0.02	0.00	0.00	0.00
	Paradise Flycatcher	*Terpsiphone viridus*	Müller 1776	0.08 ± 0.05	0.00	0.05 ± 0.04	0.02 ± 0.02	0.23 ± 0.07	0.00	0.00	0.00
Family Malaconotidae											
	Southern Boubou	*Laniarius ferrugineus*	Gmelin 1788	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04 ± 0.03
	Puffback	*Dryoscopus cubla*	Shaw 1809	0.08 ± 0.03	0.03 ± 0.03	0.53 ± 0.16	0.04 ± 0.04	0.53 ± 0.08	0.53 ± 0.40	0.13 ± 0.07	0.29 ± 0.07
Family Sturnidae											
	Blackbellied Starling	*Lamprotornis corruscus*	Nordmann 1835	0.00	0.00	0.00	0.00	0.07 ± 0.04	0.00	0.00	0.02 ± 0.02

Appendix 2 continued on next page...

Appendix 2 (continued)

Order and family	Common name	Scientific name	Date of description	Age of study site							
				10 years		14 years		18 years		Unknown age (unmined forest)	
				Edge	Core	Edge	Core	Edge	Core	Edge	Core
Family Nectariniidae											
	Whitebellied Sunbird	*Nectarinia talatala*	Smith 1836	0.13 ± 0.11	0.03 ± 0.03	0.10 ± 0.07	0.05 ± 0.03	0.00	0.07 ± 0.07	0.00	0.00
	Grey Sunbird	*Nectarinia veroxii*	Smith 1831	0.02 ± 0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Olive Sunbird	*Nectarinia olivacea*	Smith 1840	0.42 ± 0.12	0.03 ± 0.03	0.14 ± 0.07	0.03 ± 0.02	0.58 ± 0.14	0.27 ± 0.13	0.07 ± 0.07	0.21 ± 0.07
Family Zosteropidae											
	Cape White-eye	*Zosterops pallidus*	Swainson 1838	0.55 ± 0.20	0.40 ± 0.17	0.42 ± 0.14	0.17 ± 0.09	1.58 ± 0.4	0.80 ± 0.51	0.00	0.24 ± 0.09
Family Ploceidae											
	Thickbilled Weaver	*Amblyospiza albifrons*	Vigors 1831	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05 ± 0.03
	Forest Weaver	*Ploceus bicolor*	Vieillot 1819	0.32 ± 0.15	0.05 ± 0.05	0.13 ± 0.10	0.06 ± 0.04	0.58 ± 0.18	0.37 ± 0.23	0.20 ± 0.20	0.02 ± 0.02
	Spottedbacked Weaver	*Ploceus cucullatus*	Müller 1776	0.02 ± 0.02	0.12 ± 0.08	0.00	0.00	0.00	0.00	0.00	0.00
	Southern Masked Weaver	*Ploceus velatus*	Vieillot 1819	0.01 ± 0.01	0.00	0.00	0.02 ± 0.02	0.00	0.00	0.00	0.00
	Yellow Weaver	*Ploceus subaureus*	Smith 1839	0.03 ± 0.03	0.00	0.00	0.02 ± 0.02	0.00	0.00	0.00	0.02 ± 0.02
Family Estrildidae											
	Green Twinspot	*Mandingoa nitidula*	Hartlaub 1865	0.00	0.00	0.00	0.00	0.07 ± 0.04	0.00	0.00	0.00
Family Fringillidae											
	Yellow - eyed Canary	*Serinus mozambicus*	Müller 1776	0.10 ± 0.05	0.00	0.03 ± 0.03	0.00	0.07 ± 0.07	0.00	0.13 ± 0.13	0.02 ± 0.02