CHAPTER 1

"Nature, in some form, shall undoubtedly survive the next few centuries, but will we?"   (Cox & Moore 1993).

INTRODUCTION

Afforestation and habitat fragmentation

Large-scale afforestation with exotic trees such as pines, is taking place in montane grasslands throughout South Africa (Armstrong & van Hensbergen 1999). The Drakensberg escarpment of Mpumalanga constitutes one of the core areas of the South African forestry industry. It is also one of the oldest afforested areas with many plantations being older than 50 years. These activities have resulted in the large-scale fragmentation of the indigenous North-Eastern mountain sourveld (Acocks 1953) of these mountains.

In addition, authors such as Matthews et al. (1993) and Deall (1985) indicated that this area constitutes a habitat very rich in endemic plant taxa – close to a tenth of the 23 000 species of seed plants indigenous to the flora of the southern Africa subregion has been recorded from the Mpumalanga Escarpment, and a total of 81 species and intraspecific taxa are endemic or near-endemic to the region. These plants, most of which are herbaceous with low reproductive rates and with large underground structures, are not able to colonise disturbed areas rapidly and are therefore potentially threatened by forestry activities. Attempting reintroduction of rare grasslands species elsewhere (Morgan 1999), has shown no promise or guarantee of success, stressing

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that preservation and sympathetic management of remnants must be the central strategy for the conservation of species in fragmented landscapes.

In order to promote sustainable afforestation of these grasslands, a better understanding of the dynamics and interactions of the grassland patches remaining in the area, is mandatory. The forestry industry is aware of these problems and this project is in collaboration with SafCol to provide information required for a management plan for the conservation of the remaining biodiversity in the area.

Management of grassland

Because original continuous grassland on the Mpumalanga escarpment today remain mainly as fragmented patches in a matrix of unnatural landscape characteristics, natural, sporadic disturbances that are so important to the well-being of any system, are seldom allowed to influence communities in these remnant patches. Many authors (Swengel 1998, Greatorex-Davies & Sparks 1994, Welch 1998) have indicated the need for grassland management. As a result of fundamental differences between the relatively undisturbed grasslands of the past and the grassland fragments of the present, the replication of the natural influences of the past will not satisfy the conservation and management needs of the remaining grassland patches left today (Swengel 1998).

Deciding to manage grassland is, however, the easy part. Determining the optimal management regime for a particular patch, choosing from different options, answering important questions of how, when and how often to manage, complicates the challenge of conserving grassland biodiversity greatly. Grassland patches must also be

treated on individual merit, since the effects of these management options often depend on the specific environmental conditions and evolutionary history of each patch (Drechsler & Wissel 1998). To generate a successful grassland management plan for a particular fragment, the effects of each management option and the effects of the interaction between different management options should be understood and accounted for.

Comparing mechanical cutting, for instance, to grazing, highlights particular important differences. Cutting is sudden and catastrophic, rather than gradual and prolonged, it is also non-selective, and the treading and fertilising effects of grazing are absent (Morris 1981). Although some results favour cutting above grazing (Swengel 1998), and other haying above burning (Swengel 1996), there is no evidence to suggest that any one option provides optimal maintenance of biodiversity. Some evidence suggests that where management options are combined, greater conservation success transpire than is the case if any one option is implemented alone (Gross et al. 1998). It is obvious that, even when using options in combination, these options need to be controlled (Katoh et al. 1998), and often a full range of management regimes are required, if the survival of all species is to be ensured (Fensham 1998).

Grassland and fire.

Fire is widely used in the management of ecosystems worldwide, mainly because it is a relatively inexpensive option by which vegetation, and indeed all grassland ecosystem components, can be manipulated (Bond & Wilgen 1996). The greatest problem concerning fire, as a management tool is the fact that managers, researchers

and students constantly treat fire as a solitary but repeatable event, seldom accepting that not two fires are the same (Whelan 1995).

When planning management regimes, it is important to remember that a fire has the immediate characteristics of intensity, season, extent and type, and the historical characteristics of pre-fire climate, time since last fire and the historical characteristics of previous fires. Present day land use conflicts ensure that land management activities, such as prescribed burning, should equally satisfy conservation needs as well as more utilitarian objectives. This would rule out activities such as hazard reduction burning, which are typically carried out at a time of year when it is easiest to control fires.

Although at first glance a sound idea, it raises serious concerns about the potentially deleterious ecological impacts of out of season burning (Whelan 1995). Furthermore, such a fixed regime will inevitably lead to the impoverishment of diversity by repeatedly selecting for the same species at the expense of others (Bond & van Wilgen 1996). To guard against the adverse effects of a fixed fire regime, frequency, season and intensity of fires used should be varied – such variability is inevitable in nature and essential in prescribed-burning plans designed to conserve biodiversity. It has been argued that historical fire-interval distributions were largely the result of large fires, during extreme weather conditions (dry in forests and wet in arid regions). Information such as long-term weather records can consequently be most beneficial, if they can be related to specific areas burned each year (Gill & McCarthy 1998).

Grassland flora is directly and severely influenced by fire intensity, season and frequency. Changes in species composition typically occur, with some species, such as *Themeda triandra*, disappearing very rapidly when fire is excluded (Whelan 1995). In the Equadorian grass páramo, the frequency of fires determined the amount of fuel that accumulated within grass tussocks, and some plants may be unable to survive repeated burning (Ramsay & Oxley 1996). One of the important factors explaining phytomass variation in tallgrass prairie vegetation was seasonal burn type (Coppedge 1998). Their results also indicate that fire intensity can be an important factor determining species distribution patterns in the Chilean mattoral, and that at areas where fires of high intensity occurred, only vigorous resprouters would dominate, while less intense fires allowed new species to get established (Segura 1998).

Although it could be argued that all such examples are isolated cases, the conclusion that fire regime has a distinct effect on community structure in grassland vegetation, is accurate. The community being the biotic unit receiving a fire, animals are likewise influenced by fire. Montane grasshoppers in Southern Africa use hilltops as thermal refugia, and burning of these hilltops not only accentuate thermal influences, but also initially deprive grasshoppers of food (Samways 1990). Fire influences prairie butterflies differently, with generalist species being over-represented in more recently burned areas, and specialists showing the most negative effects from fire (Swengel 1996). Even though this is true, fire need not be excluded as a management option, since occasional wildfires were typically more favourable for specialist abundance than regular rotational burning (Swengel 1998).

Creating the optimal fire regime, and combining it with other management options such as mechanical cutting and grazing, to devise a sensible management plan for grassland areas that will appeal to the conservationist, land owner and government official alike, is no easy task. But with an understanding of the dynamics of grassland communities gained from the recording of appropriate data and the design of appropriate experiments, there may be a future for remnant patches of grassland in a landscape of alien and disturbed habitat.

Grassland and invasive aliens

One of the greatest threats to grassland biodiversity is the change in community structure brought about by the various exotic plants that are aggressively invading grassland areas, and seriously threatening to transform patches of diverse grassland into systems of poor stability and primary succession. Areas such as the species rich and productive riparian zones are particularly susceptible to invasion, and may act as havens, corridors and sources of exotic plant invasions for upland sites (Stohlgren et al. 1998). Trees used in forestry are especially problematic, and have severely impacted large areas of grassland by causing shifts in life-form dominance, reducing structural diversity, increasing biomass, disrupting prevailing vegetation dynamics and changing nutrient cycling patterns (Richardson 1998). The aggressively invasive nature of these trees cannot be ignored. Other groups of plants such as forbs and grasses have also proven to present problems for grassland conservationists.

The seriousness of the situation has clearly been demonstrated by the extensively positive response of grasses after removal of exotic forbs. This is only possible by aggressive management of target invaders with selected herbicides (for instance) in the early stages of exotic plant invasion (Rice & Toney 1998). Invasion of natural

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areas by highly aggressive exotic vegetation could be, in the long term, perhaps the most spectacular and important threat to rare and endangered species (Thiollay & Probst 1999). If stability and diversity of grassland systems are to be maintained, creating grassland areas free from alien components should be a primary conservation objective.

Grassland and loss of natural habitat

Destruction of natural habitats that leads to species extinction is a conservation issue of major concern worldwide. Those natural areas still remaining are more often than not found as patches in a fragmented landscape resulting from various land use conflicts. Many ecological surveys using rapid inventory techniques assume that a diverse floral component imply high diversity for the invertebrates as well (Crisp et al. 1998). This assumption may prove to be dubious, especially for grassland.

Grassland areas that are deprived of natural disturbances such as grazing and wildfires become less diverse with time if left unmanaged for long (Swengel 1998). More often than not, managing grassland can transform the invertebrate community (Swengel 1996, Morris 1981 & Swengel 1998) without much alteration of the floral component. A model for species-specific predictions of the impact of habitat fragmentation would assist greatly in conserving biodiversity (MacNally & Bennett 1997), especially since most of the grassland in the study area is affected by habitat fragmentation.

This study attempts to identify areas of particular conservation importance by looking specifically at the native plants (Chapter 2), invertebrates and birds (Chapter 3) and

using all these groups together to consider the present conservation status of the remaining grassland fragments (Chapter 4).

Fragmentation: Effects on natural populations

<u>Edges</u>

The effects of habitat edges on species diversity, species richness and ecosystem functioning in remnant patches of natural habitat are complex and difficult to interpret, and a clear pattern is yet to present itself. As the size of a habitat shrinks, so does the area free from edge effects (Gaston 1994), and these edge effects reduce the effective reserve area (Esseen & Renhorn 1998).

In Brazilian forest fragments species diversity and species richness for small mammals was shown to increase significantly as the distance from the edge increased (Stevens & Husband 1998). However, duck-nesting success was not significantly influenced by distance from edge (Pasitschniak-Arts <u>et al</u>. 1998), and in the Wog Wog Fragmentation Experiment (Margules 1992) there was no difference in species composition between inner and outer zones. Some groups such as grasshoppers have been shown to be distributed regardless of obvious landscape boundaries (Ingham & Samways 1996). To confuse matters further, so-called 'edge species' have been shown to make important contributions to the diversity patterns of remnant patches (Quinn & Robinson 1987).

Landscape boundaries (or ecotones) are described as sections of space (or time) where ecological conditions change more rapidly than in adjacent areas, leading to rapid changes in structure, function and composition of the biota (Samways 1989). Such

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ecotones have been proven to be, to some degree, important in satisfying life-cycle needs of many organisms, and are generally characterised by high biological diversity (Holland et al. 1991).

Island biogeography

With the intensification of land use conflicts and with the pressures of a growing human population, most of the natural areas over the world have been fragmented into remnant patches, or so-called 'islands'. Most of the predictions of population dynamics and ecosystem functioning of these patches have centred on the theory of island biogeography of MacArthur and Wilson (1967).

The theory predicts that insularity leads to the loss of species until the biota have 'relaxed' to a new, lower equilibrium species number appropriate for its size and degree of isolation. Fragmentation of a population into small, isolated populations leads to inbreeding depression, loss of genetic variability, accumulation of mild deleterious mutations and genetic adaptations which can all cause higher extinction rates than in 'mainland' populations (Frankham 1998). Furthermore, fragmentation of source populations could lead to decline in sink populations, without further fragmentation of the sink populations (Trine 1998), and such subpopulations' viability is the key to metapopulation stability (Drechsler & Wissel 1998). Thus, for a given structure of a natural environment, predictions can be made about the dynamics and persistence of a given metapopulation, but these predictions are complicated (Foord 1997).

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Metapopulation structure

Most natural populations have come to possess some geographic structure – species whose ranges are now composed of more or less geographic isolated patches, interconnected through patterns of gene flow, extinctions and recolonizations, form metapopulations. The general effect of population subdivision is to somewhat increase the total genetic variation in the metapopulation system, but at expense of decreasing the variation within subpopulations (Lande & Barrowclough 1987). It is therefore not only the total size of the metapopulation that is important when considering population vulnerabilities, but also the geometrical character of its distribution and the dynamical aspects of the flow of its member individuals between nodes of the network of habitable patches (Gilpin 1987).

Limitations of 'accidental' fragmentation experiments

To be able to obtain robust results from any fragmentation experiment, the experiment should be designed to incorporate solutions to the problems of size-effects and random sites effects relevant to any field experiment (Margules 1992, Van Jaarsveld et al. 1998). Adequate replication in field experiments is necessary to compensate for size-effects, and sufficient variation of fragment sizes essential to insure that random sites-effects do not affect the results significantly (Margules 1992).

Environmental variation is a major source of experimental error in any field experiment, but within planned fragmentation experiments it is possible to account for this variation by including adequate replicates of fragments of different sizes as well as suitable controls. Planned fragmentation experiments are often limited by the maximum amount of land available (Margules 1992) and often by the physiography

of the terrain (Van Jaarsveld et al. 1998). Furthermore, time puts an extensive limitation on any planned fragmentation experiment, since it takes many years for the fragments to become truly isolated. 'Accidental' fragmentation experiments frequently suffer from important design faults, including reduced opportunities for adequate replication and relevant controls. It is also seldom possible to incorporate and anticipate environmental variation, and therefore being able to draw firm inference from the results.

The mountain grassland of the escarpment region of Mpumalanga is all but destroyed. Many grassland specialists of the area such as the blue swallow are on the brink of local extinction. Many of the plant communities described by Matthews et al. (1993) are no longer evident in the study area, and although it would be ideal to design a proper fragmentation experiment, time and available grassland left is of the essence. All the grassland fragments and relevant controls left in the area (as far as practically possible) are included in this study to produce the best results possible considering time and spatial constraints.

Synthesis

Human-induced habitat fragmentation has changed the face of natural habitats all over the world. No longer the exception, it has influenced the population dynamics of most species on earth, and continue to threaten the continued existence of many animals and plants. Treating fragmented populations of species as interconnected metapopulations and realising the effects of edges, management of natural areas and invader species will be crucial when considering conservation options. Although the effects of edges, ecotones and subpopulation-interactions are vague and varied, it

can't be denied that they are real, and influence the populations of species and community structure greatly.

Indicator taxa

The whole indicator species concept has often been criticised, and said to be problematic, at best. Often there is little consensus on what the indicator is supposed to indicate, and it is never easy to decide which is the best indicator species (Simberloff 1998, McGeogh 1998). Despite this criticism, indicators and surrogates have often been used, in a number of different ecological studies.

 It is often the case that distribution data is inadequate, and, consequently, reserve selection procedures have to rely on surrogate measures for biodiversity (Wessels et al. 1999). For management of natural areas, monitoring of terrestrial arthropod indicators can provide early warnings of ecological changes, and can be used to assay the effects of further fragmentation on areas that no longer support vertebrate indicator species, and the unparalleled diversity of arthropods provides a rich data source that can improve the spatial resolution of biological inventories and hence the planning of reserve networks (Kremen et al. 1993). It is, however, important that conservation managers that use indicators realise the importance of them being region-specific (Trine 1998).

Due to a lack of complete taxonomical understanding of most arthropod groups, ecologists are often forced to use morphospecies as surrogates for species. At least some evidence suggests that this is acceptable in some environmental monitoring and conservation, in particular when estimates of richness and the assessment of turnover

guide decisions. The conservative evaluation of habitat remnants may be performed, in part by defining selected invertebrate morphospecies assemblages characteristic of those remnants (Oliver & Beattie 1996).

The insect orders Orthoptera and Coleoptera are used in this study, in particular for their diversity, ease of sampling, high mobility and high biomass (Foord 1997). Grasshoppers, especially, are good indicators for general grassland communities, are relatively sensitive to landscape disturbance and a direct indication of grass species diversity (Samways & Moore 1991). Butterflies and day-flying moths (Lepidoptera) are used in this study for their habitat specificity, strong host plant preferences, and ease to monitor over any extended period of time (Pringle et al. 1994). Butterflies have also been shown to be indicative of the diversity of insect communities, and are sometimes considered to be suitable 'process indicators' (Oostermeijer & van Swaay 1998). Other groups showing potential as bioindicators include soil dwelling Diptera (Fourz 1999), woodlice (Paoletti & Hassall 1999), ants (Lobry de Bruyn 1999) tiger beetles (Rodriguez et al. 1998), Syrphidae (Sommaggio 1999), Staphylinidae (Bohac 1999) and pollinator guilds (Kevan 1999).

Specific taxa or species can be used as ecological indicators to show the effects of environmental change (such as habitat fragmentation) on biological systems, if chosen correctly. (McGeogh 1998, Rodriguez 1998). Although charismatic vertebrates have been used extensively in the past, invertebrates are far better suited for the purpose of ecological indication. They have high population growths, short generation times and are far more habitat specific and prone to microclimatic changes, making them ideal indicators (Rodriguez 1998). Having said that, very little is known about many insect species in terms of biology and distribution, and bioindication relies completely on

species identification, supported by knowledge of the basic biological and ecological features of the organisms and landscape under study (Paoletti 1999).

Hypotheses to be tested:

1) To test whether the grassland plant communities identified by Matthews <u>et al</u>. (1992) over a very large spatial scale and by Deall (1985) over an extremely restricted spatial scale is also useful for characterising the plant communities in and around the pine plantations of the study area (Chapter 2)

2) To test to what degree the montane grassland fauna in the study area reflect the plant communities that have been identified (Chapter 3).

3) To test whether the assemblages of insects along edges of grassland fragments are in any way affected by the adjacent plantation (Chapter 4).

4) To test whether fragment size has any effect on the species richness, species diversity and assemblage structure of the grassland insects of the study area (Chapter 4).

METHODS AND PHYSICAL ENVIRONMENT

Area reconnoitering

Areas that could possibly represent grassland were identified on 1:10 000 aerial
photographs, and this information superimposed on 1: 50 000 topographical maps.
The study area is covered by six 1: 50 000 topographical maps (2430DC Ohrigstad,
2430DD Graskop, 2530BA Long Tom Pass, 2530BB Sabie, 2530BC Boshalte and
2530BD Nelspruit), stretching from the Blyde River Nature Reserve in the North, to
the Sudwala caves in the South, and from God's Window in the East to the Long Tom
Pass in the West (Figure 1). Using roads and other prominent landmarks, areas of
forestry, grassland and other land uses (such as urban areas and newly planted
plantations) were identified on the aerial photos and marked on the maps. Using these
maps, and a Garmin GPS 40, the whole area was surveyed for accessible remnant and
control plots of mountain grassland.

Distribution and number of sample plots

The number of sampling plots is usually influenced by various factors such as the
scale of the survey, environmental heterogeneity in the study area and the scale
necessary for the classification (Bredenkamp 1982, Matthews 1991). The biggest
influence on the choice and distribution of plots in this study was accessibility to the
sites, and of natural grassland not invaded by the various problematic exotic invaders
found in the area. Many of the sites identified on the aerial photographs as possible
areas of grassland are not accessible by land vehicle, and can only be reached by long
excursions on foot or by air.

The twenty-four sites, of which six can be regarded as control plots, are situated between 1130 and 1980 metres above sea level, and range from 0.5 to 500 Ha (Table 1). The forestry agencies SafCol and Mondi are responsible for most of the sites with a few under private and government ownership. Only two sites benefit from any sort of official protection. Types of management range from fire and grazing to cutting and no management at all, and are inconsistent in timing and coverage.

Geology

Information concerning geology was obtained from 1: 250 000 geological survey maps (Geological Survey 1986). The whole area belongs geologically to the Transvaal sequence, and the study sites fall into two subgroups, Chuniespoort and Wolkberg, and two formations, Black Reef and Timeball Hill. As far as the Lithology is concerned, the underlying rocks consist mainly of dolomite, lime, shale and quartzite. The Black Reef Quartzite formation varies from 0 m to 500 m thick and is a succession of clean quartzite with layers of pebbles; shale is usually present near the top, on the contact zone with the overlying dolomite. The formation is resistant to weathering and is responsible, by and large, for the shape of the cliffs and gorges throughout the area. The Chuniespoort group overlies the Black Reef and is mainly made up of dolomite, with some limestone and chert. It is, on average, 400 m thick (Matthews 1991).

The general soil patterns of the area are very complex because of the complexities in topography and differences in weathering of the different rock types. Because the area has a high rainfall, many of the soils show medium to high leaching as well as high acidity (average pH of 4.7), with the soils of the dolomite being the least acidic (Matthews 1991, 1992).

Table 1. Description of the 24 study sites in the Escarpment region of Mpumalanga. Sample plot characteristics are (from left to right): geological formation, lithology, elevation (metres above sea level), slope, fragment size (ha), aspect (degrees), distance to nearest grassland (km), max. temp. (° C), min. temp. (° C), average temperature (° C), average rainfall (mm), latitude and longitude.

Sites	Geological Formation	Lithology	Eleva-tion	Slope	Size	Aspect	Near-grass	Maxtemp	Mintemp	Avetemp	Averain	LAT	LONG
1	Timeball Hill	Shale	1980	6.5	106	180	1.25	19.2	8.9	14.1	1128	24° 50' 56"	30° 41' 07"
2	Black Reef	Quartzite	1130	11.3	1.5	225	0.875	21.8	9.5	15.6	1484	24° 53' 36"	30° 50' 05"
3	Chuniespoort	Dolomite,lime,shale	1350	6.1	1.9	225	0.1	21.9	10.3	16.1	1455	24° 57' 03"	30° 48' 54"
4	Black Reef	Quartzite	1364	4.2	3.6	135	0	22.4	11.5	16.9	1588	24° 58' 36"	30° 50' 20"
5	Chuniespoort	Dolomite,lime,shale	1260	4.9	1	225	0.5	22.9	10.2	16.6	1322	24° 57' 43"	30° 49' 18"
6	Black Reef	Quartzite	1260	5.7	210	270	0.5	22.4	11.5	16.9	1416	25° 00' 34"	30° 49' 53"
7	Timeball Hill	Shale	1270	7.2	1	135	0.1	23.1	11.5	17.3	1361	25° 04' 08"	30° 44' 54"
8	Chuniespoort	Dolomite,lime,shale	1290	17.7	0.8	135	0.1	20.9	10.7	15.8	1690	25° 03' 27"	30° 44' 16"
9	Chuniespoort	Dolomite, lime, shale	1290	19.6	0.5	135	0.1	22.8	10.6	16.7	1317	25° 02' 12"	30° 45' 39"
10	Black Reef	Quartzite	1860	8.7	34	135	1.45	20.8	10.6	15.7	1343	25° 11' 06"	30° 39' 24"
11	Timeball Hill	Shale	1860	6.5	3.8	90	1.55	21.5	9.4	15.4	1070	25° 10' 34"	30° 37' 36"
12	Timeball Hill	Shale	1800	9.1	2.1	315	1.45	20.9	10.5	15.7	1230	25° 09' 59"	30° 40' 14"
13	Timeball Hill	Shale	1530	22.2	0.9	315	2.5	21.5	10.1	15.8	1432	25° 10' 40"	30° 42' 31"
14	Timeball Hill	Shale	1200	7.9	64	45	2.1	21.6	9.8	15.7	1253	25° 12' 16"	30° 40' 26"
15	Chuniespoort	Dolomite,lime,shale	1220	9.9	15	90	4.8	23.5	10.7	17.1	1032	25° 15' 38"	30° 41' 38"
16	Chuniespoort	Dolomite,lime,shale	1380	7.4	143	315	0.325	22.3	11.6	17	1157	25° 19' 45"	30° 47' 12"
17	Black Reef	Quartzite	1260	15.4	267	360	0.325	22.7	11.7	17.2	972	25° 18' 58"	30° 47' 24"
18	Chuniespoort	Dolomite,lime,shale	1470	36	33	45	1.95	23.1	11	17	1268	25° 08' 11"	30° 45' 41"
19	Timeball Hill	Shale	1860	12.9	500	90	0	20	9.1	14.6	1268	25° 09' 49"	30° 37' 43"
20	Black Reef	Quartzite	1590	9.1	500	315	0	21	9.1	15	1700	24° 51' 17"	30° 52' 38"
21	Black Reef	Quartzite	1470	6.5	500	315	0	21.4	9.5	15.4	1627	24° 51' 31"	30° 52' 09"
22	Chuniespoort	Dolomite,lime,shale	1440	1.3	500	360	0	22.1	9.6	15.8	1293	24° 55' 33"	30° 49' 32"
23	Chuniespoort	Dolomite,lime,shale	1380	7.6	500	225	0	21.7	9.7	15.7	1320	24° 55' 27"	30° 49' 16"
24	Timeball Hill	Shale	1950	18.4	500	90	0	15.6	5.5	10.6	1497	25° 09' 23"	30° 37' 27"

Table 2. Collection dates of the eight sweep samples for Coleoptera and Orthoptera, and coinciding hand net samples for Lepidoptera and Neuroptera.

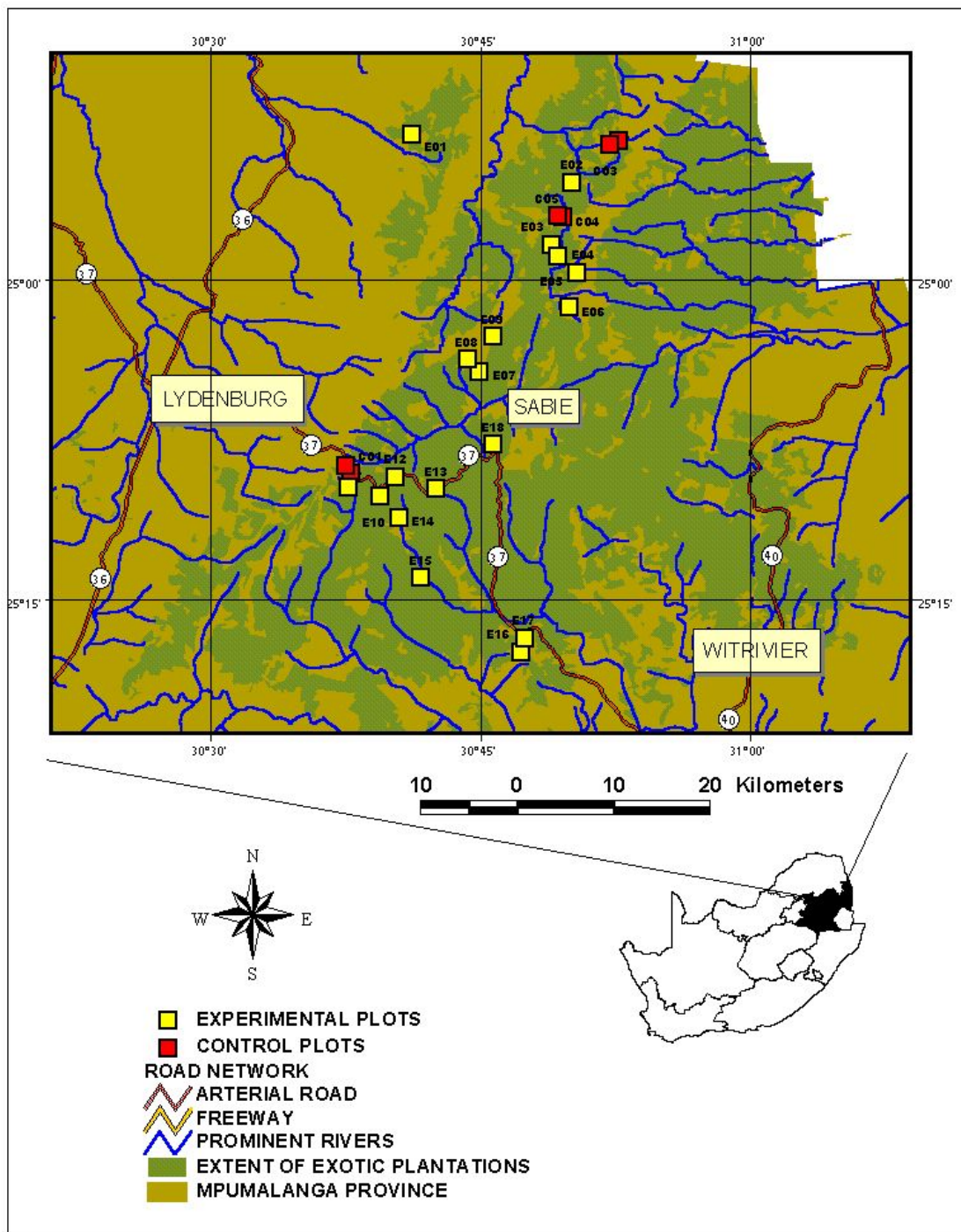| Seasons	Feb - Mrch-97		Oct - Nov-97		Feb-98		Nov-98	
Site no's	sweep 1	sweep 2	sweep 3	sweep 4	Sweep 5	Sweep 6	sweep 7	sweep 8
1	10.ii.97	26.ii.97	23.x.97	22.xi.97	6.ii.98	22.ii.98	10.xi.98	24.xi.98
2	10.ii.97	26.ii.97	23.x.97	21.xi.97	9.ii.98	20.ii.98	11.xi.98	28.xi.98
3	12.ii.97	2.iii.97	26.x.97	21.xi.97	6.ii.97	12.ii.98	11.xi.98	25.xi.98
4	12.ii.97	2.iii.97	26.x.97	14.xi.97	6.ii.97	12.ii.98	11.xi.98	25.xi.98
5	17.ii.97	2.iii.97	24.x.97	14.xi.97	6.ii.98	12.ii.98	11.xi.98	25.xi.98
6	12.ii.97	2.iii.97	26.x.97	7.xi.97	6.ii.98	15.ii.98	11.xi.98	25.xi.98
7	17.ii.97	12.iii.97	26.x.97	7.xi.97	8.ii.98	15.ii.98	11.xi.98	23.xi.98
8	17.ii.97	2.iii.97	26.x.97	7.xi.97	8.ii.98	15.ii.98	11.xi.98	23.xi.98
9	18.ii.97	2.iii.97	26.x.97	7.xi.97	8.ii.98	15.ii.98	11.xi.98	24.xi.98
10	18.ii.97	3.iii.97	1.xi.97	17.xi.97	7.ii.98	13.ii.98	6.xi.98	23.xi.98
11	18.ii.97	3.iii.97	3.xi.97	17.xi.97	7.ii.98	13.ii.98	6.xi.98	23.xi.98
12	16.ii.97	3.iii.97	1.xi.97	15.xi.97	7.ii.98	13.ii.98	12.xi.98	23.xi.98
13	16.ii.97	3.iii.97	1.xi.97	15.xi.97	7.ii.98	13.ii.98	12.xi.98	23.xi.98
14	21.ii.97	3.iii.97	30.x.97	20.xi.97	11.ii.98	25.ii.98	17.xi.98	24.xi.98
15	21.ii.97	3.iii.97	30.x.97	20.xi.97	11.ii.98	23.ii.98	17.xi.98	24.xi.98
16	21.ii.97	3.iii.97	30.x.97	20.xi.97	11.ii.98	23.ii.98	17.xi.98	24.xi.98
17	21.ii.97	3.iii.97	30.x.97	20.xi.97	11.ii.98	23.ii.98	17.xi.98	24.xi.98
18	16.ii.97	3.iii.97	1.xi.97	15.xi.97	7.ii.98	13.ii.98	12.xi.98	23.xi.98
19	16.ii.97	3.iii.97	5.xi.97	17.xi.97	7.ii.98	16.ii.98	4.xi.98	23.xi.98
20	10.ii.97	26.ii.97	23.x.97	22.xi.97	9.ii.98	20.ii.98	11.xi.98	26.xi.98
21	10.ii.97	26.ii.97	23.x.97	22.xi.97	9.ii.98	20.ii.98	10.xi.98	26.xi.98
22	12.ii.97	26.ii.97	23.x.97	21.xi.97	9.ii.98	20.ii.98	10.xi.98	26.xi.98
23	12.ii.97	26.ii.97	23.x.97	21.xi.97	9.ii.98	20.ii.98	10.xi.98	26.xi.98
24	16.ii.97	3.iii.97	5.xi.97	20.xi.97	7.ii.98	16.ii.98	4.xi.98	23.xi.98

Figure 1. Map of the study area.

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