

**BIOTIC INDICATORS OF GRASSLAND CONDITION IN
KWAZULU-NATAL, WITH MANAGEMENT
RECOMMENDATIONS**

By

RICHARD GRANT KINVIG

**Submitted in partial fulfilment of the
requirements of the degree of
Doctor of Philosophy,
in the School of Biological and Conservation Sciences
University of KwaZulu-Natal
Pietermaritzburg
2005**

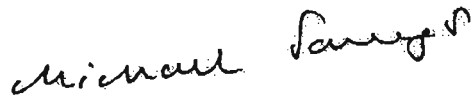
DECLARATION

This study was carried out in the School of Biological and Conservation Science, University of KwaZulu-Natal, Pietermaritzburg from 2001 to 2005 under the supervision of Professor Michael J Samways. This is the author's original work and has not been submitted in any form to another university. Where the work of others has been used, it is duly acknowledged.

I declare that the above statement is correct



Richard Grant Kirvig



Michael J Samways

ABSTRACT

The South African grassland biome is disappearing rapidly through advancing development and change in agricultural land use. One of the most threatened grassland types, Midlands Mistbelt, in the KwaZulu-Natal Midlands is an extremely diverse and home to many endemic species across an array of taxa. Three taxa, namely, grasses, grasshoppers and butterflies represent various trophic levels, which are important to the functioning of the grasslands. Ten grasslands were sampled by walking ten fifty metre transects for a twelve-month period. The grasslands were selected as they represented a range of management practices and varying environmental conditions. Using Indicator Species Analysis (ISA) twenty-two species of grasshopper were identified as indicators of environmental variables and management practices. The abundances of the various species indicated the intensity of the management regimes or disturbances. Using the twenty-two grasshopper species abundances and a three hundred point sampling assessment of the grasses creates an assessment tool that can rapidly appraise the management of the grassland, but due to lack of data for other taxa, cannot assess whether management practices for the focal taxa create congruent results for non-focal taxa. Two of the three taxa proved to be good indicators of grassland health, whilst the third, butterflies were ineffectual, due to low abundance and richness. From the results it was concluded that burning was taking place too frequently, and required a reduction to every four years, as this would improve butterfly richness and abundance, and increase abundance of endemic and flightless grasshopper species. A rotational grazing system needs to be implemented at sites where continual grazing takes place, wildlife or livestock, impacts on the grassland condition and species diversity. Increasing habitat heterogeneity increases species diversity, and allows later successional species to be included in the grasshopper assemblage. Management of the grasslands in the KwaZulu-Natal Midlands needs to be more responsive and adaptive. In addition, small fragment management needs to be intensified to provide a range of habitats and refugia that will suit all species. This study advocates the use of grasshoppers and grasses as suitable biotic indicators of grasslands in the KwaZulu-Natal Midlands.

CONTENTS	PAGE
ABSTRACT	i
CHAPTER 1	1
INTRODUCTION	1
1.1. Threats to grasslands	2
1.2. Indicator species	6
1.3. Invertebrates as indicators	8
1.4. Orthoptera as indicators	8
1.5. Potential threats to orthopteran biodiversity	11
1.6. Lepidoptera as indicators	12
1.7. Potential threats to butterflies	13
1.8. Grasses as indicators	15
1.9. Indicator taxa choices	15
1.10. Study aims	18
CHAPTER 2	20
SITES, MATERIALS AND METHODS	20
2.1. Study sites	20
2.2. Sampling	28
2.2.1. Grasses	28
2.2.2. Grasshoppers	28
2.2.3. Butterflies	30
2.2.4. Environmental variables	30
2.3. Statistical analysis	31
2.3.1. Data transformation	31
2.3.2. Canonical ordination	32
2.3.3. Cluster Analysis and evenness indices	33
2.3.4. Analysis of Variance (ANOVA)	34
2.3.5. Pearson Correlations	34
2.3.6. Mantels Test	34
2.3.7. Multiple Response Permutation Procedures (MRPP)	35
2.3.8. Indicator Species Analysis (ISA)	36

CHAPTER 3	37
RESULTS	37
3.1. Grass species composition and abundance	37
3.2. Veld condition scores	39
3.3. Forbaceous species composition and abundance	39
3.4. Abundance across species	41
3.4.1. Grasshoppers	41
3.4.2. Butterflies	44
3.5. Richness and abundance across sites	46
3.5.1. Grasshoppers	46
3.5.2. Butterflies	52
3.6. Endemic species	56
3.6.1. Grasshoppers	56
3.6.2. Butterflies	58
3.7. Seasonal variation	59
3.7.1. Grasshoppers	59
3.7.2. Individual grasshopper responses to seasonality	68
3.7.3. Butterflies	72
3.8. Correspondence analysis	80
3.8.1. Grasshopper species composition	80
3.8.2. Butterfly species composition	83
3.8.3. Environmental variables	84
3.9. Species responses to environmental variables	85
3.9.1. Grasshoppers	85
3.9.2. Butterflies	87
3.10. Species responses to nominal variables	88
3.10.1. Grasshoppers	88
3.10.2. Butterflies	89
3.11. Grasshopper feeding guilds	91
3.12. Assemblage responses to grazing regimes	91
3.12.1. Grasshoppers	91
3.12.2. Butterflies	92
3.13. Grasshopper responses to forestry	92
CHAPTER 4	94
INDICATORS OF GRASSLAND CONDITIONS	94
4.1. Surrogate subsets of grasshoppers	94
4.2. Surrogate subsets for butterflies	96
4.3. Correlations between grasshoppers, butterflies & grasses	97
4.4. Grasshopper surrogate subset	98
4.5. Grasshopper assemblage responses to environmental categories	99

4.6. Grasshopper indicator species.....	99
4.6.1. Topographic indicators	100
1. Aspect	100
4.6.2. Grasshoppers indicative of four vegetation categories	100
1. Proportion of forbaceous plant material.....	100
2. Vegetation height.....	101
3. Grasshopper species representative of grassland veld condition.....	101
4. Grassland type	102
4.6.3. Management Indicators	103
1. Grazing.....	103
2. Biennially burnt grasslands.....	104
3. Disturbed grasslands	104
4. Season in which the grasslands were burnt	104
4.7. Butterfly assemblage differences.....	105
4.7.1. Topographic indicators	106
1. Aspect	106
4.7.2. Vegetative indicators	106
4.7.3. Management indicators	107
1. Grazing	107
2. Disturbance.....	107
3. Biennially burnt grasslands.....	107
4. Season in which the grassland was burnt.....	107
 CHAPTER 5	 109
DISCUSSION	109
5.1. Grassland types	109
5.2. Impact of abiotic variables on grasshopper assemblages.....	111
5.2.1. Slope orientation.....	111
5.2.2. Elevation	112
5.3. Impact of biotic variables on grasshopper assemblages.....	113
5.3.1. Vegetation structure and architecture	113
5.3.2. Habitat heterogeneity and grasshoppers	115
5.4. Landscape level and the impacts associated with man	118
5.4.1. Defoliation	118
1 Livestock grazing.....	120
2. Wildlife grazing.....	123
5.4.2. Disturbance: its effect on the grasshopper assemblage.....	125
1. Burning.....	126
2. Species responses to burning	130
3. Season of burning.....	131
4. Mowing.....	133
5.5. Sampling period, grasshopper richness and abundance.....	136
5.6. Dominance of certain species.....	137
5.7. Significance of elevated grasshopper abundance.....	137

CHAPTER 6	139
CONSERVATION BIOLOGY AND ECOLOGICAL ASSESSMENT	139
6.1. <i>Flagships, umbrella and keystone species</i>	141
6.2. <i>Ecosystem management</i>	144
6.3. <i>Rapid assessment</i>	145
6.3.1. <i>Surrogate choice</i>	149
1. <i>Families</i>	149
2. <i>Functional groups</i>	149
6.3.2. <i>Choice of grasshopper species as indicators</i>	150
1. <i>Topography</i>	151
2. <i>Vegetative characteristics</i>	152
3. <i>Indicators of management</i>	154
Grazing.....	154
Season in which the grassland was burnt.....	154
Biennial burning versus other forms of defoliation.....	155
Disturbance.....	155
6.3.3. <i>Choice of butterfly species as indicators</i>	158
 CHAPTER 7	 159
CONCLUSIONS & MANAGEMENT RECOMMENDATIONS ...	159
7.1. <i>Conclusions</i>	159
7.1.1. <i>Responses to structural heterogeneity</i>	159
7.1.2. <i>Responses to grazing</i>	159
7.1.3. <i>Responses to fire</i>	160
7.1.4. <i>Responses to mowing</i>	160
7.1.5. <i>Assessment of grasslands</i>	161
7.2. <i>Management recommendations</i>	161
7.2.1. <i>Has past management affected the species assemblage?</i>	161
7.2.2. <i>Grassland management to improve biodiversity</i>	162
7.3. <i>Future research</i>	166
 ACKNOWLEDGEMENTS	 167
 REFERENCES	 168

LIST OF FIGURES

PAGE

Figure 1 (a). The study site area in the province of KZN, SA.....	21
Figure 1 (b). The ten study sites.....	21
Figure 2 (a) to (j). Photographic representations of the ten study sites that were sampled over a twelve-month period in the KwaZulu-Natal Midlands.....	25 - 28
Figure 3. A histogram of grasshopper abundance over the 12-month study period for all 10 sites. (Species represented by hatched bars are endemic). Refer to Table 6 for grasshopper species codes.....	44
Figure 4. Butterfly species recorded across ten sites for the 12-month study period. (Species represented by hatched bars endemic). Refer to Table 7 for butterfly species codes.....	45
Figure 5 (a) to (j). Abundance of grasshopper species across each of the 10 study sites over the whole study period. Hatched bars represent Midlands Mistbelt and full bars Highland Sourveld. Refer to Table 6 for grasshopper species codes.....	47 - 49
Figure 6. Grasshopper similarity across sites using Bray-Curtis Cluster Analysis. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....	49
Figure 7. A bar chart of species richness for the ten sites sampled during the 12-month study period. (The hatched bars represent Highland Sourveld grasslands and the black bars Midlands Mistbelt). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....	52
Figure 8 (a) to (j). Butterfly abundance per species for the whole study period. (Hatched bars represent Midlands Mistbelt grasslands, Black, Highland Sourveld). Refer to Table 7 for butterfly species codes.....	54-55
Figure 9. Butterfly similarity using Bray-Curtis Cluster Analysis across sites. (Data were combined and log transformed). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....	56
Figure 10. A bar chart of species endemism for the ten sites sampled during the 12-month study period. (The black bars represent Highland Sourveld grasslands and the hatched bars Midlands Mistbelt). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....	56
Figure 11. A bar chart of butterfly species endemism for the ten sites sampled during the whole study period. (The blue bars represent Highland Sourveld grasslands and the red bars Midlands Mistbelt). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....	58
Figure 12. Grasshopper similarity across months using a Bray-Curtis Cluster Analysis for all sites (Combined site data).....	59

- Figure 13. Correspondence Analysis (CA) of months for all sites using grasshopper assemblages. (Only centroid values for each month were plotted). (Only species with greater than 20% of their variance explained are plotted). Refer to Table 6 for grasshopper species codes. Axis 1 and 2 Eigen values are 0.611 and 0.320 respectively.....61
- Figure 14. Eigenvector scores on the first axis of a Correspondence Analysis (CA) versus months to show grasshopper species compositional change over one year for all sites.....62
- Figure 15. (a) to (l). Grasshopper species recorded across all of the ten sites for each month of the study period. (Summer months are represented by hatched bars and winter by full bars). (Refer to Table 6 for grasshopper species codes).....62 - 66
- Figure 16. An analysis of variance comparing the summer and winter grasshopper species richness for the twelve-month study period.....66
- Figure 17. Species richness of grasshoppers during the whole study period.....67
- Figure 18 (a) to (n). Seasonal variation in species that have a cumulative fit of greater than 20% of their variation explained for Axis 1 and 2. All data have been log-transformed.....69 - 72
- Figure 19. Butterfly similarity across months using a Bray-Curtis Cluster Analysis. (Data were combined and log transformed).....73
- Figure 20. Correspondence Analysis (CA) of month distribution according to the butterfly species for the 10 sites.....74
- Figure 21. Eigenvector scores versus month for all the sites for butterflies (each dot represents a site).....75
- Figure 22 (a) to (l). Butterfly abundances for species in each month of sampling. (Summer months indicated by hatched bars and winter by full bars). (Refer to Table 7 for butterfly species codes).....75 - 78
- Figure 23. Correspondence Analysis (CA) for grasshopper species composition for all sites across all months. (Data were combined and log transformed). Refer to Table 6 for grasshopper species codes. The Eigen values for Axis 1 and 2 were 0.177 and 0.142 respectively.....81
- Figure 24. The Correspondence Analysis (CA) for the ten sites across the 12-month study period. (Only centroid values are shown for each month). (J= January, F= February, M= March, A= April, MY= May, JE= June, JY= July, AU= August, S= September, O= October, N= November, D= December).....81 - 82
- Figure 25. Correspondence Analysis (CA) of butterflies for the whole study period. Refer to Table 7 for butterfly species codes. The Eigen values were 0.227 and 0.168 for Axis 1 and 2 respectively.....83
- Figure 26. Partial Correspondence Analysis (PCA) of the environmental variables measured illustrating colinearity between the environmental variables. (Abbreviations as per Table 19).....84
- Figure 27. Canonical Correspondence Analysis (CCA) for environmental variables against the sum of the species (Species with 50% variance explained plotted). The combined axes were significant (F= 1.301 and p= 0.045) (Abbreviations as per Table 19). Refer to Table 6 for grasshopper species codes.....86

- Figure 28. Canonical Correspondence Analysis (CCA) of butterflies versus environmental variables.** The combined axes were not significant ($F= 1.309$ and $p= 0.105$) (Only species with $>50\%$ variance explained were plotted). (Grass= overall grass %; Min T= Minimum temperature recorded at each site during the whole study period). (Refer to Table 7 for butterfly species codes).....88
- Figure 29. Canonical Correspondence Analysis (CCA) for nominal variables against species** (Species with 50% variance explained plotted). Combined axes were not significant ($F= 0.809$ and $p= 0.290$). (Abbreviations as per Table 19). (Refer to Table 6 for grasshopper species codes).....89
- Figure 30. Canonical Correspondence Analysis (CCA) of butterfly species versus nominal variables.** (Only species with $>50\%$ of their variance explained were plotted). The combined axes were not significant ($F= 1.226$ and $p= 0.110$) (LG= livestock grazed; WG= wildlife grazed; B2001= burnt in 2001; B2002= burnt in 2002). (Refer to Table 7 for butterfly species codes).....90
- Figure 31. A box and whisker plot diagram comparing species richness and type of grazing** (LG= livestock, UG= Ungrazed and WG= wildlife grazed).....91
- Figure 32. Total grasshopper species richness plotted against family richness and number of endemics for the ten study sites.** (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....94
- Figure 33. Surrogate grasshopper abundance for the ten sites compared to total grasshopper abundance.** (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)..... 95
- Figure 34. Surrogate grasshopper richness for the ten sites compared to total grasshopper richness.** (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)..... 95
- Figure 35. Total butterfly species richness plotted against family richness and endemics for the ten study sites.** (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)..... 96
- Figure 36. Surrogate butterfly abundance for the ten sites compared to total butterfly abundance.** (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).....96
- Figure 37. Surrogate butterfly richness for the ten sites compared to total butterfly richness.** (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)..... 97

LIST OF TABLES

PAGE

Table 1. Sites, grassland type, elevation above sea level and the GPS grid reference for the centre of each site.....	20
Table 2. Sites, there locality to an urban centre, the reasons why they were chosen, there management regimes(grazing, burning) and the dominant grass species recorded at each site.	24
Table 3. The grassland type (Highland Sourveld (H.S.) and Midlands Mistbelt (M.M.)), VC1KEY and VC2Bench are Key and Benchmark scores respectively for grasslands, environmental variables measured (Average temperature, Relative humidity, Minimum temperature, Slope orientation, Grass, Rock, Forbaceous plants, Bareground and Invasive (alien or indigenous) species as a total percentage of the grassland) and average vegetation height (cm).	25
Table 4. Grass species abundances recorded during February 2001 across the ten study sites sampled. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)	38
Table 5. A list of the most commonly encountered forbaceous plants and authorities (Germishuizen & Meyer, 2003) recorded during the study period at the ten study sites.	40
Table 6. Grasshopper species recorded during the study period, the code used in analyses, frequency of encounters for each species, the mean of species abundance, the mean per transect and the maximum number of individuals encountered at any one transect.	41
Table 7. Butterfly species recorded during the study period. The code used in analysis, frequency of encounters for each species, the mean of species abundance, the mean per transect and the maximum number of individuals encountered at any one transect are given.	44
Table 8. Grasshopper diversity indices, evenness and species richness for each of the 10 sites between February 2001 and October 2002 (Data combined and log transformed). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).	46
Table 9. A presence-absence table of grasshopper species for the ten sites over the study period (* denotes presence). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).	50
Table 10. Presence-absence of butterfly species for the ten sites over the whole study period (* denotes presence). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).	52
Table 11. Butterfly diversity indices, Shannon-Wiener, evenness and species richness for each of the ten sites over the whole study period (Data combined and log transformed). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).	53

- Table 12.** A presence absence table of endemic grasshopper species for the ten sites over the 12-month study period (* denotes presence). (Abundance measure, Low = (2-30), Medium = (60-100) and High = (140-647)). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2). 57
- Table 13.** A presence absence table of endemic butterfly species for the ten sites over the 12-month study period (* denotes presence). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2). 58
- Table 14.** A presence absence table of grasshopper species during each month of the study period (* denotes presence). 60
- Table 15.** Grasshopper species richness when the two sampling periods were compared summer = October to March, winter = April to September. (***) Denotes significant difference at $p < 0.05$. 67
- Table 16.** Grasshopper diversity indices, evenness and species richness for each of the 12 months of the study period (data log transformed). 67
- Table 17.** Butterfly diversity indices, evenness and species richness for each of the 12 months of the study period (data were log transformed). 75
- Table 18.** A presence absence table of butterfly species for the 12-month study period (* denotes presence). 79
- Table 19.** The codes and explanations of the codes used in the Canonical Correspondence Analysis, and variables omitted as their affects are mirrored by other variables (* indicates omitted). 85
- Table 20.** The cumulative fit of the species, for the whole study period, versus the environmental variables, as a fraction of the variance, for grasshopper species with greater than 50% variance. Refer to Table 6 for grasshopper species codes. 86
- Table 21.** A one-way ANOVA of feeding guilds versus Veld Type (Highland Sourveld and Midlands Mistbelt) for grasshopper species across the ten study sites. 91
- Table 22.** A one-way ANOVA of species richness and species evenness versus land-use type (Grazing) for grasshopper species across the ten study sites. 91
- Table 23.** The LSD post hoc test was undertaken to show any significant differences between grazing strategies and species richness (5% significance denoted by *) (LG= livestock, UG= Ungrazed and WG= wildlife grazed). 92
- Table 24.** Butterfly richness and abundance when compared across the three different grazing types. 92
- Table 25.** The abundance of grasshoppers when comparing forestry (plantations) with the indigenous grassland. Figures in parentheses indicate the number of species recorded. 93
- Table 26.** Pearson's correlation values when comparing like categories between the butterfly and grasshopper assemblage. * = Significant at the 0.01 level. 98
- Table 27.** Mantel's test correlations between total and surrogate grasshopper species richness, and total surrogate species abundance. Observed Z greater than average Z from the randomized runs indicates a positive association. 98

Table 28. The number of samples (transects that need to be walked) needed to achieve the relevant confidence intervals (C.I.), for total species assemblage and surrogate grasshopper species assemblage. 98

Table 29. The results of Multiple Response Permutation Procedures (MRPP) for nine categories using the grasshopper abundance data collected across the ten sites for the twelve-month study period. (n.s. = Not significant, * = Significant $p = 0.05$, ** = Significant $p = 0.01$ and *** = Significant $p < 0.01$). 99

Table 30. Eight species were indicative of slope orientation (Aspect). Aspect = Warm or Cool Slopes. (Warm = -, Cool = +). Refer to Table 6 for grasshopper species codes. 100

Table 31. 21 species were indicative of various vegetative assessments. Forbcon= % forbaceous material (Low = +, High = -), Veg height = vegetation height (cm) (Short = +, Tall = -), VCKey= Veld Condition scores (Poor = -, Good= +) and Grasstype = Grassland Type (Highland Sourveld or Midlands Mistbelt) (Highland Sourveld = +, Midlands Mistbelt = -). ++, -- denotes a very strong indicator. Refer to Table 6 for grasshopper species codes.....103

Table 32. 24 species were indicative of various management regimes. Grazing = Livestock or Wildlife grazed (Grazing = +, Not grazed = -), Biennial = Biennially burnt (Biennial = -, Other = +), Disturbed = Disturbance (Over-utilisation) (Low = -, High = +) and Season of burn = winter or spring (winter = -, spring = +). -- denotes very strong indicator. Refer to Table 6 for grasshopper species codes. 105

Table 33. The results of Multiple Response Permutation Procedures (MRPP) for nine categories using the butterfly abundance data collected across the ten sites for the twelve-month study period. (n.s. = Not significant and * = Significant $p = 0.05$).. 106

Table 34. Eight species were indicative of slope orientation (Aspect). ASPECT = Warm or Cool Slopes. (WARM = -, COOL = +). Table 7 for butterfly species codes. 106

Table 35. Three butterfly species were indicative of various vegetative assessments. Forbcon= % forbaceous material (Low = +, High = -), Veg height = vegetation height (cm) (Short = +, Tall = -), VCKey= Veld Condition scores (Poor = -, Good= +) and Grasstype = Grassland Type (Highland Sourveld or Midlands Mistbelt) (Highland Sourveld = +, Midlands Mistbelt = -). Table 7 for butterfly species codes. 106

Table 36. 10 species were indicative of various management regimes. Grazing = Livestock or Wildlife grazed (Grazing = +, Not grazed = -), Biennial = Biennially burnt (Biennial = -, Other = +), Disturbed = Disturbance (Over-utilisation) (Low = -, High = +) and Season of burn = winter or spring (winter = -, spring = +). Table 7 for butterfly species codes. 108

Table 37. The 22 species of grasshopper utilised to assess the various management regimes and responses to environmental disturbance. (The abundance of each of the species will need to be recorded in order to assess the intensity of the impact or perturbation). 153

Table 38. Grasses and grasshopper species characteristic of grasslands that fall into one of three categories, i.e. poor, moderate or excellent. The grasses are sampled using a random three hundred point sampling technique. The grasshoppers are sampled by walking ten fifty metre transects through the grassland. The period in which sampling is undertaken is important because of the species richness and distribution across the species. The best time to sample grasshoppers is in February and March (greatest species richness), with the best sampling time for grasses being February, when inflorescences are present. It is important to note that this is not a definitive guide to grassland condition, but is based on the findings of this study and directs us towards certain conclusions. 157

**"IT IS NOT THE STRONGEST OF THE SPECIES THAT SURVIVE,
NOR THE MOST INTELLIGENT, BUT THE ONE MOST RESPONSIVE
TO CHANGE."**

CHARLES DARWIN

CHAPTER 1

INTRODUCTION

There is consensus within the scientific community that the 'current massive degradation of habitat and extinction of many of the Earth's biota is unprecedented and is taking place on a catastrophically short timescale' (Novacek & Cleland, 2001). Between 1989 and 2009 it is estimated that one million species of plant and animal will be exterminated due to human interference and transformation of natural ecosystems (Reid & Miller, 1989). Over the next three hundred years it is expected that between 100 000 and 500 000 species will become extinct (Mawdsley & Stork, 1995). The emerging field of conservation biology is a response to this rapid collapse of biological integrity and to the decline of biodiversity (Meffe & Carroll, 1994). One of the most important tasks is to identify and follow the changes of biota in space and time to prevent degradation and further loss of biodiversity (Báldi & Kisbenedek, 1997; Cincotta *et al.*, 2000). To do this, surveys and comparisons of different sites must be undertaken and monitoring programmes need to be developed to maintain the sites with ecological integrity (Báldi & Kisbenedek, 1997).

Insects and all their interactions with the world around them are major components of biodiversity. These animals are under threat as much as other biota (Samways, 1994). There are however, two major impediments to insect conservation.

Firstly, there is the question of taxonomy, known as the taxonomic impediment (New, 1984), and, more recently as the taxonomic challenge (Samways, 2002a). It is difficult to conserve what we do not know, and the fact that more than 80% of species remain unknown (Hawksworth & Kalin-Arroyo, 1995) means that there is a massive void in our knowledge. The second is the perceptual impediment. Insects are viewed by many people as nuisances, but less than one percent of species are actually pests (Pimentel *et al.*, 1992), with their roles in ecosystem processes not being wholly appreciated (Samways, 1997). Sotherton and Self (2000) documented that advancements in farming practices have contributed to the impoverishment of many invertebrate groups on arable land. The greatest problem facing conservation

biology, in southern Africa is the lack of taxonomic knowledge concerning insects and the lack of trained personnel to obtain that knowledge (Scholtz & Chown, 1993). Therefore insect conservation is at a very low level in southern Africa at a research and application level. The only species being considered or identified as endangered are, almost without exception, visible species that have been fortuitously identified as such (Scholtz & Chown, 1993).

It is therefore essential to increase our knowledge of insects and their behavioural intricacies so as to understand how and why they are important in ecosystems. Insects are not only important as a food source for many of the larger more recognizable vertebrates, but also play a major role in ecosystem functioning as they often are the largest contributors to overall ecosystem biomass (Gandar, 1982a; Gangwere, *et al.*, 1997). A prime example of this relationship is a complex series of interactions between mammals, insects and their host plants in mopane savanna, with insects exhibiting a key role (Bryant *et al.*, 1991).

1.1. Threats to grasslands

One of the most important drivers in environmental change is the increased anthropogenic use of land (Sala *et al.*, 2001; Neke & du Plessis, 2004). Under the pressures of a rapidly increasing human population, and the resultant large-scale expansion of agriculture, industry and housing, reduces large expanses of previously undisturbed natural vegetation, into fragments surrounded by a transformed matrix of alien vegetation or development.

Grasslands are one of the most disturbed and transformed areas in the world (Hannah *et al.*, 1994; Tarboton, 1997a). They are transformed more easily than other existing land uses, as there is less effort required to transform them. Effort can be broken down into two distinct categories, human resources and cost effectiveness of the transformation.

Grasslands constitute 70% of all land surface (Fuhlendorf & Engle, 2001). Grasslands in the northern hemisphere alone once covered some 600 million ha, with very few natural remnants left today. The grassland biome of South Africa

consists of 25 veld types and covers an area of 349,174 km² (Neke & du Plessis, 2004) or 16.5% of the land mass vegetative cover (Dawson, 1991). Based on total habitat loss, degree of fragmentation and estimation of future threat, the South African grassland biome has been identified as critically endangered (Olsen & Dinerstein, 1998; Reyers *et al.*, 2001; Foord *et al.*, 2002) and the biome most in need of conservation attention in the country (Rebelo, 1997). Grasslands are prevalent in most of the higher elevation areas of South Africa, particularly in the Eastern Cape, Mpumalanga and KwaZulu-Natal Midlands. The differentiation into the major plant communities is generally related to climate and elevational change (O'Connor & Bredenkamp, 1997). Due to good soils and relatively high and consistent rainfall much of this area has been converted to agricultural and forest lands (Macdonald, 1989). The grasslands that have not been transformed are characteristically managed for livestock production (Holechek *et al.*, 1998). This matrix needs to be managed so that some or all of the original components of the ecosystem are preserved. This semi-natural matrix dominates many regions in KwaZulu-Natal and the world and may contain most of the world's biological diversity (Pimentel *et al.*, 1992).

Moist Midlands Mistbelt is the most threatened veld type in KwaZulu-Natal (Scott-Shaw, 2002). 131 000 ha (34%) out of 381 000 ha remains untransformed by cultivation and development. 10 000 ha (8%) remains as natural grassland in fragments larger than 32 ha (Scott-Shaw, 2002). At least 60% is in an impoverished condition i.e. it has a greater than 70% loss of species and can play no part in conservation in the short- or medium-term future (Scott-Shaw, 2002). At present, approximately 4000 ha (1%) remains, with the potential for conservation in the short term. Only 928 ha (0.25%) of grasslands are protected in six reserves, with three reserves having less than ten hectares of grassland between them. Most of the grassland destruction has come from establishment of forestry plantations throughout this area.

Pine trees are alien to South Africa (Mirov, 1967). As aliens they are expected to have a detrimental impact on the indigenous biota and ecosystem functioning. Afforestation changes the soil, hydrology, habitat structure, microenvironment, food resources and ecological processes of the landscape (Armstrong & van

Hensbergen, 1996). The plantations play a major role in the reduction of underground water with each pine species using 15 litres of water, and a Eucalypt species 150 litres per day (Pers. Comm. G. Zaloumis). There is considerable debate in South Africa on the impacts of forestry on biodiversity (Smith, 1974; Johns, 1993; Cellier, 1994; Armstrong & van Hensbergen, 1996; Armstrong *et al.*, 1996; Spellerberg & Sawyer, 1996; Pott, 1997; Armstrong *et al.*, 1998).

However, the commercial forestry industry and associated processing industries form a significant part of the South African economy (Kruger *et al.*, 1995). In 1979 the forestry and associated industries contributed 7.70 billion Rand to the Gross Domestic Product (GDP), by 1992 the contribution of the timber industry constituted 1.2% of the GDP of South Africa (Forestry South Africa, 1993). The Rand value was c.a. 9.57 billion, an increase of c.a. 20% in 13 years. By 2002 this figure had increased to 14,28 billion Rand an increase in Rand terms of 33% (Forestry South Africa, 2004).

In 1979 the amount of land that was under timber was 1,096,455 hectares (ha), by 1992 this figure had increased to 1,307,207 ha and at present in 2002 stands at 1,351,402 ha. The percentage land usage increase over the last ten years has been a year on year one percent increment (Forestry South Africa, 2004). This industry is important to the poverty-stricken, local human communities, which it provides with employment. The importance of forestry to the GDP of South Africa is tangible, but its growth comes at the sacrifice of land, especially grasslands situated in the Eastern Cape, KwaZulu-Natal Midlands and Mpumalanga (formerly the Eastern Transvaal) (Armstrong & van Hensbergen, 1997).

Grasslands therefore need to be viewed as a form of 'natural' capital or 'environmental' asset, as with indigenous forest, in order to compare them with other land-uses, like forestry. Grasslands are primarily involved in the production of grazing lands for livestock and conservation areas for biodiversity, which is an international tourism draw card. Grasslands also play a role in ecosystem functioning, the protection of local watersheds and supporting the economic livelihood of indigenous people, who utilise the grasslands for thatching grass, grazing of livestock and the collecting of plants for medicinal purposes. Therefore,

grasslands play an important role in the long-term economic welfare of South Africa, as they contain a complex disturbance regime composed of frequent large- and small-scale disturbances that interact with interannual climate variation to affect spatial and temporal dynamics of species assemblages (Collins, 1987; Coffin & Lauenroth, 1988; Day & Detling, 1990; Bragg, 1995; Frank *et al.*, 1998; Knapp *et al.*, 1999). When grassland is being conserved, an opportunity cost is being incurred, through the loss of potential income gained from changing a land-use. If grasslands are to be an efficient means of holding onto wealth, they must yield a rate of return that is comparable or greater than that of other land-uses. Therefore, we need to consider all avenues of wealth that grasslands afford us.

There are many hidden values and services that are not considered when evaluating grasslands. These services would include recreation, tourism and education. There are also less tangible values that are lost, for example, the foreclosing of future uses of the goods provided by the grasslands, as well as knowing that such a system can no longer be shared with future generations. These values are paramount and need serious consideration when assessing grasslands, both economically and environmentally.

As an individual, a species does not exist in isolation, but is always part of a larger, more complex ecological web, in which the different organisms interact with one another as predators, prey, competitors or mutualists (Didham *et al.*, 1996; Memmott, 1999). Disruptions to these intricate relationships, may go unnoticed because of the survival ability of one or other of the web members, or they disappear altogether before man has had time to study and assess the relationships.

Understanding species diversity in local assemblages requires knowledge of processes acting at larger spatial scales, including determinants of regional species richness and spatial turnover of species (Caley & Schluter, 1997; Sax & Gaines 2003). The general factors that influence species diversity and the ecological web in terrestrial ecosystems include: climate (Currie, 1991), habitat structure and productivity (Pianka, 1966, Tilman, 1982; Morris, 1990, Rozenweig & Abramsky, 1993; York, 1999), habitat isolation and habitat area (MacArthur & Wilson, 1967; Niemelä *et al.*, 1993b; Spence *et al.*, 1996; Beaudry *et al.*, 1997) and habitat

fragmentation (Kearns *et al.*, 1998; Fuhlendorf *et al.*, 2002). All of these factors are being increasingly influenced by human undertakings and activities. With the increased pressure on natural systems, areas for conservation are becoming smaller, in turn increasing the risk of local and regional extinctions of species (Saunders *et al.*, 1991; Groom & Schumaker 1993; Hanski, 1994; Allen *et al.*, 1996; Armstrong & van Hensbergen, 1996; Latchininsky 1998; Andrieu-Ponel & Ponel, 1999).

1.2. Indicator species

Many conservation decisions in the present day are made at the large geographical scale, where indicator taxa are useful to explain patterns in biodiversity, because these patterns usually are the product of only a few factors, origination and extinction (Cracraft, 1992; Rozenweig, 1995). Conversely at smaller regional scales, biodiversity patterns are the product of these same factors plus numerous additional factors such as immigration and emigration (Gaston & Blackburn, 1995). At the micro-scale, numerous additional factors compound upon the factors at the larger scales, making resultant patterns less likely to be shared by different taxa (Pearson & Carroll, 1998). However, Pearson & Carroll (1998) demonstrated that tiger beetles (Cicindelidae) were representative of butterfly species richness even at the smallest of scales. Many authors have shown that indicator taxa are able to provide general trends for the various taxa that would be found within a regional area (Wettstien & Schmid, 1999; Foord *et al.*, 2002).

The use of indicator species has been promoted because of the impossibility of monitoring all species and habitats (Caro & O'Doherty, 1999; Lindenmayer *et al.*, 2000; Soberón *et al.*, 2000; Taylor & Doran, 2001). It has been suggested that the basis for insect indicator selection is often merely based on favoured or convenient taxa (Soulé & Kohm, 1989; Woiwood & Thomas, 1993; Williams & Gaston, 1994; McGeoch, 1998; Andersen, 1999). Other indicator selections may take into account the functional role that insects play within the ecological web and their benefits to man (e.g. predators, pests or pollinators) and this is the basis for their selection. Many authors have put forward criteria that an indicator taxon should fit, like a lock and key in order to make them worthy of the title, indicator taxon. The selection

criteria for indicators of environmental health are numerous. Below are six criteria that have been selected and suggested by several authors (Noss, 1990; Goldsmith, 1991; Spellerberg, 1991; Pearson, 1994; New *et al.*, 1995; Hamer *et al.*, 1997; McGeoch, 1998).

The indicator must be:

1. Sensitive to change (sensitive to environmental change)
2. Widely distributed
3. Easily and cost effectively measurable, collectable and identifiable (they must have a stable taxonomy)
4. Able to differentiate between natural and anthropogenic variations
5. Relevant to ecological phenomena and surrogates for other taxa
6. Economically important

Many authors have only utilised a single taxon for diversity estimates (Greenslade & Greenslade, 1987; Andersen, 1990; Churchill, 1997; Pearson and Carroll, 1998). These estimates are often used to assume that variation in the indicator taxon is representative of the variation in unrepresented taxa (Colwell & Coddington, 1994; Reid, 1998). Although this may be true in some cases, recent work has suggested that diversity patterns vary greatly across taxa, and that management practices based solely on one taxon may not safeguard or predict the diversity of others (Abbott, 1974; Kremen, 1992; Prendergast *et al.*, 1993; Báldi & Kisbenedek, 1994; Launer & Murphy, 1994; Holl, 1995; Thomas, 1995, Shapiro, 1996; Abensperg-Traun *et al.*, 1997; Cranston & Trueman, 1997; Lawton *et al.*, 1998; Niemelä & Baur, 1998; Oliver *et al.*, 1998; Reid, 1998; van Jaarsveld *et al.*, 1998; Reyers *et al.*, 2002a). It is therefore necessary for us to sample a suite of taxa in order to try and reconcile any potential differences that may occur across taxa, i.e. the "shopping basket" approach (Hammond, 1994).

Therefore it is of paramount importance that a multiple taxa approach needs to be taken so as not to misguide management requirements and regimes (McGeoch, 1998, Wettstien & Schmid, 1999; Kotze & Samways, 2001).

1.3. Invertebrates as indicators

Invertebrate fauna form a major component of ecosystem biodiversity. They fulfill critical ecological roles (Yen & Butcher, 1997). They are crucial in processes of pollination, seed dispersal, soil aeration and turnover, the breakdown of organic matter and nutrient recycling (Taylor & Doran, 2001). Invertebrates form the basis of many food webs and ecological interactions, promote soil fertility, and provide mechanisms of biological control. They may also hold many economic and medicinal benefits, and particular species are of cultural, ethical, educational, recreational and aesthetic importance. Below are five reasons why invertebrates have been considered useful indicators (Yen & Butcher, 1997):

1. They are the 'glue and building blocks' of terrestrial ecosystems (Janzen, 1987; Giller, 1996; Bohac, 1999)
2. They are ecologically and functionally important
3. There are large numbers of species providing a diverse range from which to choose
4. Many species are habitat specific
5. They are numerically predominant (Holloway & Stork, 1991)

Invertebrates often exhibit smaller distribution ranges (Solem & McKenzie, 1991; Ponder *et al.*, 1994) and divide habitat on a finer scale when compared with vertebrates. Hence, they may require habitat management or reservation prescriptions at a finer level to achieve ecological sustainability than for vertebrates (Taylor & Doran, 2001). This is an important facet of ecosystem management as invertebrates contribute heavily to the biomass of ecosystems (McGeoch, 1998). Invertebrates being such a diverse group and demonstrating different species responses to disturbance (Lawton *et al.*, 1998; Ghazoul & Hellier, 2000; Lawes *et al.*, 2005) makes invertebrates potentially good indicator species.

1.4. Orthoptera as indicators

In some articles where desirable qualities have been specified for a viable indicator group, the habitat of prime concern to the author is tropical forest (Brown, 1991;

Sutton & Collins, 1991). This habitat is typically devoid in a diverse Acrididoid assemblage as they are usually associated with more open habitats and therefore may not fit the criteria that have been laid out. Some authors therefore question the suitability of Orthoptera to act as an indicator species. However, grasshoppers are important in many ecological processes (Gandar, 1980; Samways, 1997), are a particularly valuable food resource in semi-arid systems (Mullie & Kieth, 1993) and are exceptionally common in grassland systems, the focal habitat of this study. In addition, grasshoppers are remarkably sensitive to plant species composition (Joern, 1983), plant morphological characteristics (Fielding *et al.*, 2001), plant succession (Chambers & Samways, 1998), microclimate and predator-free space (Otte & Joern, 1975), they are remarkably good indicators of habitat change.

Green (1999) has shown that South African species amount to some 553 described species. If species yet to be described were included this would raise the total to approximately 600 species. South Africa is remarkable both for its species richness and for the high number of endemic species it contains (Brown, 1974; Rentz, 1978; Johnsen, 1985, 1987).

In comparison to most other African countries, South African grasshopper taxonomy is relatively well known (Green, 1998). A current estimate has shown that when comparing East Africa (Kenya, Uganda, Tanzania, Ethiopia and Somalia) to South Africa, South Africa has twice as many grasshopper species per unit area (Green, 1998). This species richness is exceptional and contrary to general trends, which dictate that species richness is significantly higher in the tropics (Gaston & Blackburn, 1996).

It has been postulated that the reason why South Africa has so many grasshopper endemics (47%) is that the southwestern region is biogeographically isolated and has resulted in many species radiations, namely in the Pneumoridae, Lentulidae, Akicerinae and Lithidiinae. Coupled with this is the fact that there were no Pleistocene ice sheets and long periods of relative geological stability (Samways, 1995). This region appears to be richest in endemic genera, although montane grasslands farther east are also rich in endemics (Armstrong & van Hensbergen, 1997; Foord *et al.*, 2002). Montane grasslands being rich in endemics make for a

strong case in the conservation of these grasslands, as well as making Orthoptera an ideal taxon for study, due to the high number of endemics, ease of capture and high abundances.

Apart from diversity, and endemism, grasshoppers are a major if not dominant group of herbivorous insects (Gangwere *et al.*, 1997) often contributing half or more of the total arthropod biomass in the grass layer (Gillon, 1983). Some Orthoptera are keystone ecosystem components (Quinn *et al.*, 1993). In the South African situation grasshoppers constitute 93% and 76% of above ground phytophagous insect biomass for nutrient rich *Acacia* savanna and nutrient poor *Burkea* savanna respectively (Gandar, 1982b). Gandar (1983) showed that grasshoppers consumed 11.3% of above ground grass production and 5.5% of above ground forb production. Stebaev (1970) mirrored these findings while working on the Russian steppes. This implies that Orthoptera are both abundant and fundamental to nutrient recycling.

Although grasshopper assemblages can be strongly influenced by 'top-down' factors (Rowe-Rowe & Lowry, 1982), especially bird predation (Bock *et al.*, 1992; Belovsky & Slade, 1993) they are primarily controlled by 'bottom-up' factors (resources) (Isely, 1938, Joern, 1979; Gandar, 1982a; Gangwere *et al.*, 1997).

This makes grasshoppers particularly sensitive to land management practices (Samways & Moore, 1991; Kemp, 1992; Porter & Redak, 1996), especially when related to disturbance of the grass layer, by grazing (Jepson-Innes & Bock, 1989; Quinn & Walgenbach, 1990; Miller & Onsager, 1991; Rivers-Moore & Samways, 1996; Chambers & Samways, 1998; Onsager, 2000) and fire (Gandar, 1982a; Evans, 1988, Bock & Bock, 1991; Porter & Redak, 1997). Grasshoppers have also played an important role in the conservation of natural areas and preservation of habitats (Rentz & Weissman, 1981; Devoka & Schmidt, 2000; Kati *et al.*, 2004). Their local level abundance is able to represent landscape or regional abundance (Kemp *et al.*, 1990; Sergeev, 1997), and their relative abundance can be a sensitive indicator of land-usage (Bei-Bienko, 1970; Port & Thompson, 1980; van Wingerden *et al.*, 1991a; Prendini *et al.*, 1996; Lockwood, 1997; Samways & Sergeev, 1997) and plant structure (Joern, 1982; van Wingerden *et al.*, 1991b, 1992).

Grasshoppers are well represented by many families in the grasslands of South Africa (Green, 1998) and in his opinion would perform admirably as an indicator group in most South African habitats. Several authors, in the South African context, (e.g. Armstrong & van Hensbergen, 1996; Foord *et al.*, 2002) have successfully utilised grasshoppers as indicators of grassland biodiversity. Stewart and Brown (1995) have successfully used Orthoptera species diversity as an indication of the state of recovery of areas in the Karoo that had previously been sprayed with pesticides to control *Locustana pardalina* Walker. However, at this point it is important not to assume that grasshoppers are the perfect umbrella taxon for all aspects of biodiversity (Lawton *et al.*, 1998; Samways, 1999).

1.5. Potential threats to orthopteran biodiversity

Invertebrates are often sensitive to small environmental changes and even relatively minor degradation can result in loss of species (Cherrill & Brown, 1990; Collins & Thomas, 1991; Samways, 1993). Habitat fragmentation leads to many populations being reduced in size and therefore vulnerable to local extinction (Samways, 1997). However, Fielding and Brusven (1993b) showed the reciprocal is true as well with certain species having population outbreaks. Habitat fragmentation and resultant extinction are particularly true for wingless species of limited vagility (e.g. Lentulidae) and to certain species where one sex is flightless (e.g. most Pneumoridae and Porthetinae). Wright (1993) suggests that this is the case for insect fauna in the Fynbos biome of South Africa and has drawn attention to this threat. Afforestation can have two affects, 1) it can fragment habitat, and, 2) it destroys montane grassland habitats, with its characteristic faunal assemblages of Orthoptera (Armstrong & van Hensbergen, 1997).

Local peoples throughout Africa utilise larger species of Orthoptera as a food source, but this subsistence use is not a serious threat. Cognisance of growing human populations must be taken and that these larger species may be faced with increased demand. Currently, grasshoppers are of no commercial value, but are seen as more of a pest than something of commercial value. In the past, *Locustana pardalina* Walker posed a major threat to grazing land in the Karoo biome, and therefore resulted in wide spread pesticide spraying programs, which in many cases

affected non-target Orthoptera as well as general insect biodiversity (Scholtz & Chown 1993, Stewart, 1998). However, with on going studies into the field of pest control many authors are prescribing other methods, which are more environmentally friendly, and have less far reaching consequences (Latchininsky 1998; Onsager, 2000).

The last major threat to Orthoptera is alteration of agricultural practices. Barker (1985) showed that overgrazing by cattle in the Kalahari environment has modified orthopteran faunal composition, with a reduction in faunal diversity mostly owing to the loss of graminicolous species.

The conservation of the vast majority of Orthoptera will depend principally on how landscapes are conserved, with management, or how they are preserved, without it. Their populations may then be viewed as functional, unnamed components of the ecosystems. Many species will be ecologically redundant, i.e. their absence will not change substantially the physiognomy or processes of the host ecosystems (Samways, 1997). However, there are others that are keystone species (Quinn *et al.*, 1993; Chase, 1996), especially in grasslands (Joern & Gaines, 1990; Kisbenedek, 1995; Chase, 1996). Therefore in order to ensure the survival of as many species within the orthopteran assemblage as possible ensuring the survival of processes, by maintaining intact as many large landscapes as possible (Samways, 1994, 1997; Bridgewater, 1996; Samways & Sergeev, 1997; Sergeev, 1998).

1.6. *Lepidoptera* as indicators

Several studies have shown how butterfly assemblage structure and diversity change along a gradient of human disturbance (Erhardt, 1985; Leps & Spitzer, 1990; Spitzer *et al.*, 1993; Blair & Launer, 1997; Hamer *et al.*, 1997; Wood & Gillman, 1998; Pryke & Samways, 2003).

Butterflies are thought to be the most useful and suitable insects as indirect measures of environmental change, because of their high sensitivity to local weather, light levels, host plant specificity and structural changes (Erhlich *et al.*,

1972; Opler & Krizek, 1984; Weiss *et al.*, 1987; Thomas 1991; Pollard & Yates, 1993; Hill *et al.*, 1995; Robertson *et al.*, 1995; Swengel, 1996a & b, Zschokke, *et al.*, 2000; Simonson *et al.*, 2001). A rider here, however, is that all these studies have been undertaken outside of South Africa.

Butterflies are one of the few invertebrate taxa for which accurate distributional, life history and taxonomic data exist (Brown, 1991; Beccaloni & Gaston, 1995; Kremen, 1994). The taxonomy and life histories of most South African species are well documented (Henning *et al.*, 1997), and the adults are generally easily identifiable in the field. In addition, adult butterflies are important pollinators in grassland systems (Burd, 1994; Oostermeijer & Swaay, 1998). In the South African context Field (2002) showed that c.a. 30% of forbs that were studied in KwaZulu-Natal Midlands grassland were pollinated by Lepidoptera with one species, relying solely on this taxa for pollination. Due to their diverse life histories, they are able to reflect changes in the vegetation complex and thus act as an indicator of various anthropogenic or climatic perturbations.

The combination of these factors make diurnal butterflies an ideal indicator of human influence on patterns of community diversity and structure.

1.7. Potential threats to butterflies

The two greatest threats to butterfly diversity are: habitat fragmentation (Thomas, 1991; Brown & Frietas, 2000) and habitat loss (Kitahara & Sei, 2001). With the rapid development of afforestation and the change from natural grazing lands to high intensity pasture grazing, with many grasslands in the KwaZulu-Natal Midlands becoming highly fragmented or lost to agriculture or forestry. It is therefore imperative that remaining grassland be conserved and the plant species required as larval host plants be studied and protected within these areas. One method of conserving butterfly species is by the development of linkages between grassland habitats (Pryke & Samways, 2003). This allows for dispersal as well as gene flow between source and sink populations as well as allowing species to investigate new areas for colonization or utilisation.

In addition to habitat change and fragmentation, are climate change and over-utilisation and/or collection of butterfly species. Severe drying out caused by anthropogenic climate change, changes in water levels as well as elevated temperatures have caused shifts in species distributions as well as extinctions. Evidence is accumulating that some mobile generalist butterflies (Dennis, 1993; Parmesan, 1996, Parmesan *et al.*, 1999; Warren *et al.*, 2001) that live in the northern hemisphere, are showing a pole ward shift in geographical ranges. Excessively long submergence increases mortality of butterflies (Joy & Pullin 1999; Webb & Pullin, 1998). The extinction of the British Large copper butterfly *Lycaena dispar dispar* and decline of the Heath fritillary *Mellicta athalia* are likely to herald changes in insect diversity in wetlands throughout the world, and that localized, habitat specialists need to be watched very carefully as early responders of possible permanent change.

Butterflies, by virtue of their bright colours, delicate form, large size and association with flowers, are among the few insects that charm peoples' hearts (Kellert 1986). This has led, for example, to removal by tourists of 100 000 *Panaxia quadripunctaria* moths per generation in the Valley of Butterflies, Rhodes, Greece (Petanidou *et al.*, 1991). Like so many aspects of conservation, over collecting must be put in perspective and on a rational, non-emotive level. For butterflies at least, which include the most collected of all insects, New (1997) points out that the adverse effects of collecting are probably far less than that of habitat change and that simple bans on collecting play only a minor role, if any, in conservation. However, it is essential that collecting be monitored carefully because certain species with small total populations, and which may be slow breeders, may be susceptible to over collecting. Nevertheless, we must be sensitive to the fact that for certain species over collecting has caused extinction. The British large copper butterfly *Lycaena dispar dispar* appears to have been collected out of existence by 1848 (Duffey, 1968). For the 33 species of butterfly listed under the United States Endangered Species Act, 30% are threatened from over-collecting. Where by-products of lepidopteran species are utilised, as is the case with the indigenous African silk moth, it is essential to establish levels of sustainable utilisation (Veldtman *et al.*, 2002).

1.8. Grasses as indicators

Grasses have been utilised for many years as indicators of grassland health (Acock's 1988; Camp & Hardy, 1999; Tainton & Camp; 1999). Grasses are the major primary producers in a grassland system and are sensitive to disturbance, particularly to over utilisation (Acock's, 1988). Certain species are indicative of varying degrees of change within the system. Grasses in South Africa have been well documented and are easy to identify, when in flower (van Oudtshoorn, 1992). The potential threats to grass species are over utilisation and loss of habitat, which are common problems within the KwaZulu-Natal Midlands of South Africa and around the world.

1.9. Indicator taxa choices

Many people may question the use of these three taxa as opposed to using other well-known taxa, which have been proved successful indicators of ecosystem integrity. In the case of ants, carabids and amphipods their collection is highly labour intensive. When sampling ants and carabids in the grassland situation, the best method of sampling is using pitfall traps (Majer, 1980; Jaganyi, 1998; Kotze & Samways, 2001). Other methods include the removal of turfs, which is highly destructive as vacuum netting is inadequate for sampling Coleoptera (Morris & Rispin, 1987). In addition, Whitford *et al.*, (1999) reported that ant species composition, richness and abundance were unchanged across a gradient of grazing intensities. The aim of this study was to be able to rapidly assess the study taxa. This was achieved using visual assessment in conjunction with sweep-netting. The use of pitfall traps requires setting up the traps and returning to empty them. This process requires the researcher to return many times to check and refill the traps.

The traps also require frequent servicing, because of disturbance caused by monkeys, which pull the traps out of the ground, making them ineffectual. Traps that are placed along mole runs would also be dug out and they would need to be moved and replaced.

In this study most of the grasslands were grazed either by livestock (Cattle) or by wildlife. These animals would crush traps when walking over them, again causing the trap to lose its worth (Jaganyi, 1998). In addition smaller animals such as rabbits would dig around the traps, which led to discrepancies in the flushness of the trap mouth and surrounding ground affecting the efficiency of the trap. Related to setting up the trap, was the type of soil in which it was placed. Many soils when they are exposed to less water and hot conditions dry out and crack, again affecting the levels of soil when compared to the trap opening.

In the KwaZulu-Natal situation, the best sampling period is summer, which is when the highest rainfall is recorded (Kotze & Samways, 2001; Jaganyi, 1998). The exposed pitfall traps can often become flooded, and the chemical mixture diluted, which prevented the ants and carabids from dying when coming into contact with the chemical mixture, and allowing them to escape. In addition, certain traps could be less prone to flooding than other traps, thus influencing their total recorded capture. Many pitfall traps are also 'raided' by predatory species, which utilise the focal taxa captured in the pitfall traps as a food resource (Pers. Comm. S.L. Bourquin).

When comparing species abundance estimates across habitats, pitfall capture data must be utilised with caution (Mitchell, 1963; Greenslade, 1964; Niemelä *et al.*, 1990; Bieringer & Zulka, 2003). It has been reported that, even though pitfall catches reflect activity and density of the invertebrates (Theile, 1977; Luff, 1982, 1986; Topping & Sunderland, 1992), they are influenced by factors such as temperature and moisture (Ericson, 1979; Honek, 1988), surrounding vegetation (Greenslade, 1964), materials used in the trap construction (Luff, 1975), preservative used in the trap (Wagge, 1985), number, size, shape and arrangement of traps (Orbtel, 1971; Adis, 1979; Niemelä *et al.*, 1986b), biology of the species like seasonal activity rhythms and behaviour (Luff, 1986; Spence & Niemelä, 1994). All these factors have the potential to influence the total capture rate and sampling success.

Finally the issue of time and expertise needs to be raised, as a rapid assessment, suggested by the name needs to be done quickly and cost effectively. Using pitfall traps requires the samples to be taken back to the laboratory where they are identified. In many cases the samples need to be dissected (Jaganyi, 1998) for

successful identification to genus and species level, with many specimens not being able to go beyond the level of genus. This type of dissection requires expensive equipment, such as stereomicroscopes and a wealth of knowledge in taxonomy. Many authors have also indicated the advantage of identification down to species, which is not possible in many cases due to the high abundances and similarities between species. The reason for rapid assessments is that time and money are two important variables, both being rare commodities.

In this study both qualitative and quantitative data were required to answer the questions posed, therefore, this form of sampling would not be beneficial, as it is difficult to quantify the abundance of certain species when only being able to identify species to the genus level or assign the species to a morphospecies or Recognisable Taxonomic Unit (RTU) category.

Taxa such as Millipedes, were excluded from the potential sampling pool as they need an expert to identify them, and in South Africa only one taxonomist is able to identify specimens to species level. Other taxa such as the Collembola are cryptic taxa that are not considered user friendly, as identification to species level is difficult. The sampling of birds and small mammals was undertaken in the preliminary stages of the study, but were excluded from the study as they were sampling at a much coarser scale (i.e. the landscape level) as opposed to the grasshoppers and butterflies, which were predominantly restricted to one habitat (i.e. point sampling). The aim of the study was to be able to get a measure that would rank grasslands according to the species that utilise the grassland for all facets of their life history. Having achieved this, recommendations could then be made as to what avenues to follow and how best to deal with the creation of island grasslands in a sea of forestry and agricultural development. Birds and small mammals would be better suited to answering questions about fragmentation as they utilise larger home ranges and different habitats for the various components that make up their life history. The creation of the landscape mosaic would alter their perception and utilisation of the landscape. The different components that comprise the landscape would then act as differential filters (Ingham & Samways, 1996), either excluding the species or altering their behaviour.

The use of geophytes would also prove to be an impediment to sampling effort, as they need flowers for identification. These specific plants only flower for a very short space of time and may fall outside the window of regular sampling (Pers. Comm. S. Johnson).

Beyond the logistical advantages of using these three taxa, they represent a breadth of ecological trophic levels, including primary producers (grasses) and primary consumers herbivores/nectivores, (butterflies) and grasshoppers, with grasshoppers contributing the most to the consumption of vegetation within the study system, and the butterflies supplementing this consumption at a very low level. In addition, these three taxa also represent a range of vagility. The grasses are of very limited dispersal as are the grasshoppers, with a few exceptions, notably the larger more robust winged species (*Ornithacris cyanae* (Stoll) and *Cyrtacanthacris aeruginosa* (Stoll)) and the butterflies, which show wide local dispersal capabilities.

1.10. Study aims

The general hypothesis is how do species and species assemblages differ between various grassland types, with the grasslands being separated on old agriculturally based distinctions, such as, productivity, grass species composition and geology. Are the differences in species assemblage related to the structure of the microhabitat or the position of the sites in relation to slope orientation, topography and elevation (macrohabitat)? Which of the management practices studied (i.e. grazing, mowing and burning) or combination thereof provides the best species representation across the three study taxa. The above goals were chosen to elucidate which aspects, if any played an important role in the local grasshopper and butterfly assemblage patterns and distribution. In turn these findings will contribute to conservation planning and management practices utilised by farmers. The results obtained will provide the first link in the chain of developing a terrestrial rapid assessment technique that will be utilised to assess grasslands in the Kwa-Zulu Natal Midlands using a suite of species. These species respond either positively or negatively to management practices or environmental perturbations. Monitoring of this suite of species will allow grassland management to adapt in response to

environmental or anthropogenic stressors, and not detract from the grasslands productivity.

CHAPTER 2

SITES, MATERIALS AND METHODS

2.1. Study sites

The ten study sites (Fig. 1a & b) were in the KwaZulu-Natal Midlands, South Africa and were representative of a range of elevations as well as different levels of disturbance (human, machinery or livestock). Two types of grassland were sampled, namely Moist Midlands Mistbelt (BRG 5) (Tainton & Camp, 1999) and Moist Highland Sourveld (BRG 8) (Hardy, 1999). The two grassland types were chosen on the grounds that they represented the most common grassland types in the KwaZulu-Natal Midlands (Table 1). In addition, the Moist Midlands Mistbelt grassland is the most frequently fragmented and altered grassland type in South Africa, and faces the imminent danger of disappearing completely. Site selection was based on the above criteria and reinforced by factors such as their status (i.e. Natural Heritage Site), the management regime imposed upon them, the grazing regime and quality of the grassland (Table 2).

Table 1. Sites, grassland type, elevation above sea level and the GPS grid reference for the centre of each site

Site	Abbreviation	Grassland type	Elevation (m) (asl)	Grid reference
Coleford	C	Highland Sourveld	1526 m	29° 57'S 29° 28'E
Goodhope 1	GH1	Midlands Mistbelt	1380 m	29° 40'S 29° 58'E
Goodhope 2	GH2	Highland Sourveld	1498 m	29° 38'S 29° 57'E
Himeville	H	Highland Sourveld	1544 m	29° 45'S 29° 31'E
Karkloof 1	KK1	Midlands Mistbelt	1119 m	29° 23'S 30° 17'E
Karkloof 2	KK2	Midlands Mistbelt	1125 m	29° 24'S 30° 16'E
Linwood	L	Highland Sourveld	1627 m	29° 35'S 30° 04'E
Stirling	S	Highland Sourveld	1474 m	29° 35'S 30° 08'E
Wahroonga 1	WW1	Midlands Mistbelt	1422 m	29° 36'S 30° 07'E
Wahroonga 2	WW2	Midlands Mistbelt	1466 m	29° 36'S 30° 08'E

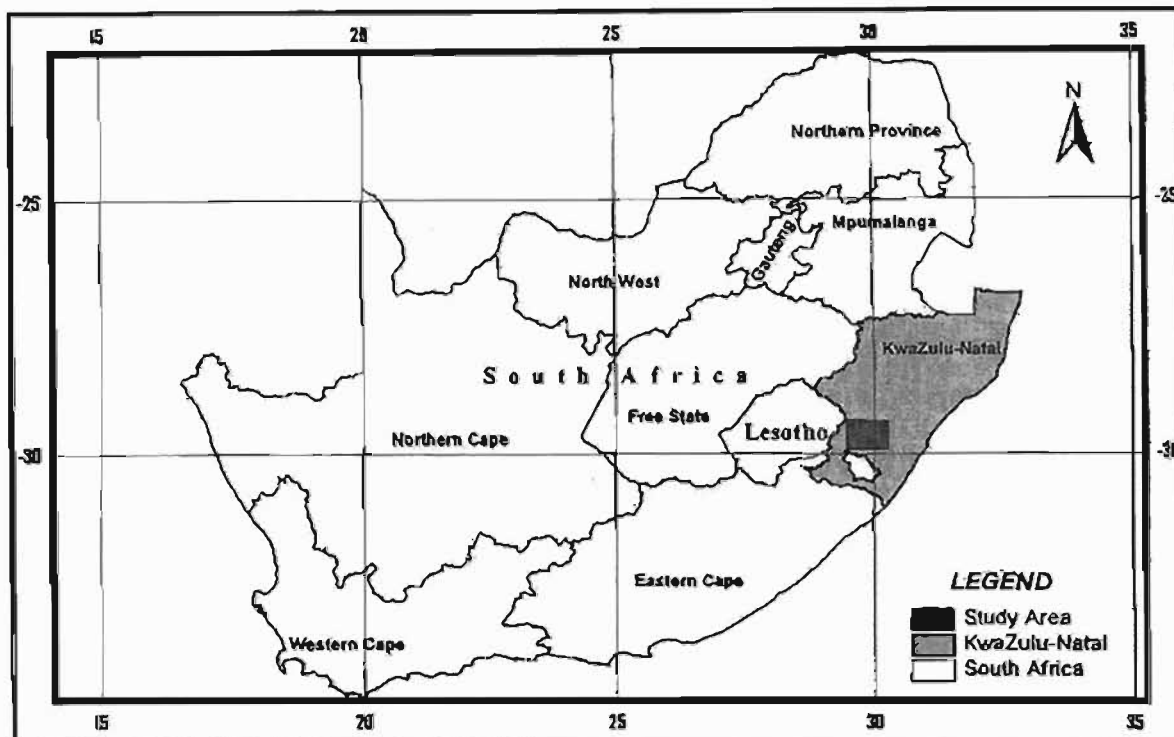


Figure 1a) The study site area in the province of KZN, SA

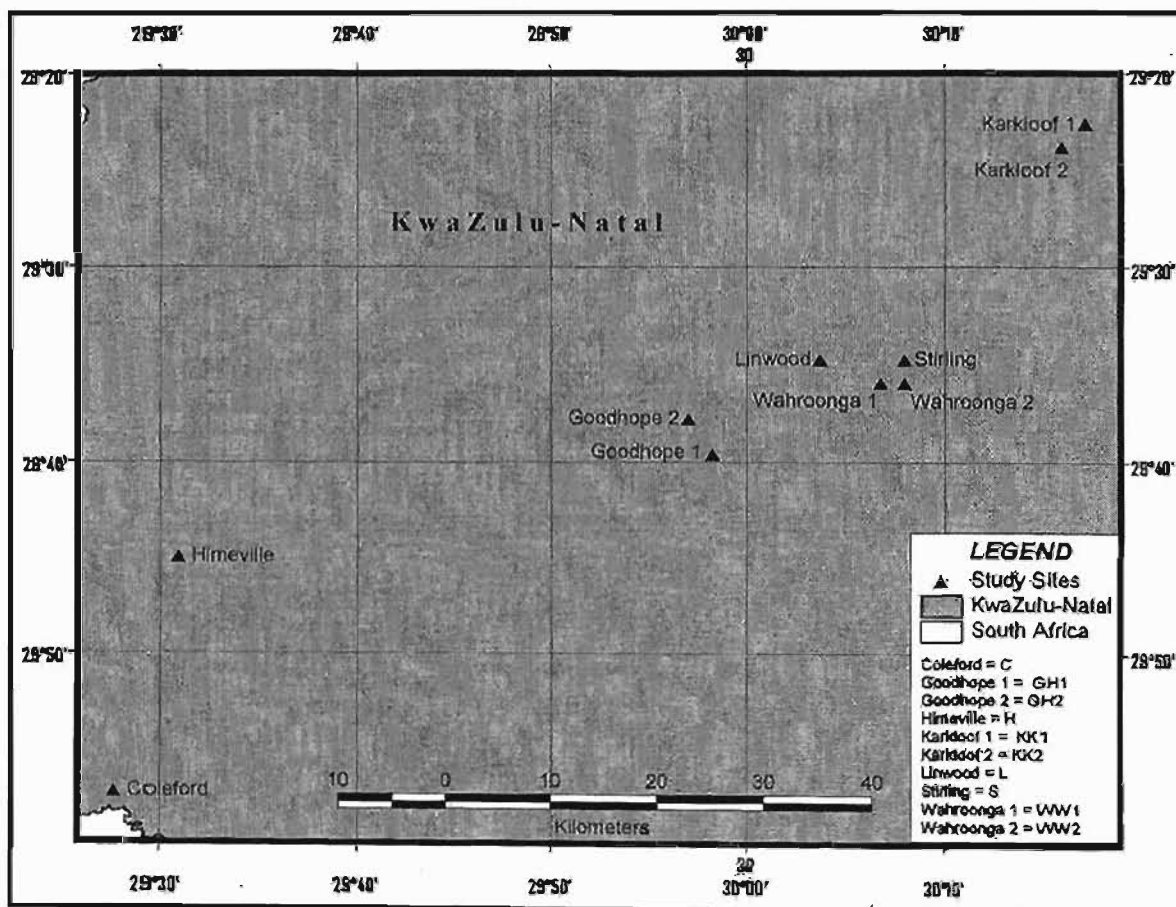


Figure 1b) The ten study sites

THE IDENTIFIERS OF GRASSLAND COMMUNITIES IN KWAZULU-NATAL AND THE ASSOCIATED ECOSYSTEM SERVICES

Table 2. Sites, their locality to an urban centre, the reasons why they were chosen, their management regimes (grazing, burning) and the dominant grass species recorded at each site

Site	Locality	Reasons for choice	Grazing	Burning	Dominant grass species
C	45km south of Underberg	Example of pristine grassland (Pers. Comm. T. O'Connor)	Wildlife*	Biennially	<i>Themeda triandra</i> Forssk.
GH 1	10km east north east of Boston at Goodhope estate	Heterogeneous grassland with adjoining wetland	Livestock**	Biennially	<i>Themeda triandra</i> Forssk., <i>Eragrostis curvula</i> (Schrad.)
GH 2	Adjoins Goodhope estate	Higher elevation, heterogeneous grassland	Livestock	Biennially	<i>Themeda triandra</i> Forssk.
H	10km north of Underberg	Example of pristine grassland (Pers. Comm. T. O'Connor)	Wildlife	Biennially	<i>Alloperis semialata</i> Nees, <i>Heteropogon contortus</i> (L.), <i>Tristachya leucothrix</i> Nees
KK 1	12 km north of Howick	Rehabilitated grassland	Absent	Biennially	<i>Aristida junciformis</i> Trin. & Rupr.
KK 2	12 km north of Howick	Part of a Natural Heritage Site	Absent	Biennially	<i>Themeda triandra</i> Forssk.
L	16km west of Howick	Highest elevation, opposite a natural forest fragment	Absent	Biennially	<i>Themeda triandra</i> Forssk., <i>Monocymbium cerisiiforme</i> Stapf, <i>Eragrostis racemosa</i> (Thun.)
S	18km west of Howick	Intensively utilised grassland by livestock	Livestock	Biennially	<i>Miscanthus capensis</i> (Nees)
WW 1	15km west of Howick	<i>O. ariadne</i> known to occur at this site***	Wildlife	Triennially	<i>Hyparrhenia hirta</i> (L.)
WW 2	15km west of Howick	This area is mowed annually	Absent	Never	<i>Themeda triandra</i> Forssk.

*Wildlife= indigenous herbivores, Black Wildebeest (C, H), Blesbuck (C, H), Mountain Reedbuck (WW 1), Grey Duiker (WW1).

**Livestock= Cattle in all cases

*** *O. ariadne*= *Orachrysops ariadne* (Karkloof Blue Butterfly) a highly localised, rare and vulnerable (D2) butterfly species.

Table 3. The grassland type (Highland Sourveld (H.S.) and Midlands Mistbelt (M.M.)), VC1KEY and VC2Bench are Key and Benchmark scores respectively for grasslands, environmental variables measured (Average temperature, Relative humidity, Minimum temperature, Slope orientation, Grass, Rock, Forbaceous plants, Bareground and Invasive (alien or indigenous) species as a total percentage of the grassland) and average vegetation height (cm)

Site	V/TYPE	VC1Key	VC2Bench	TEMP °C	RH%	Min T°C	Slope	%Grass	%Forb	%Rock	%Bare	%Invasive	AVVEGH (cm)
C	H.S	1.35	1.26	28.23	49.67	20.5	W	85	14	1	0	0	45
GH1	M.M	0.56	0.73	28.78	53.00	21.0	S	72	22	0	3	3	60
GH2	H.S	0.85	0.94	26.87	52.50	19.0	S	75	15	7	3	0	35
H	H.S	0.81	0.83	28.00	53.67	21.0	W	74	16	0	10	0	65
KK1	M.M	0.37	0.43	25.46	58.08	19.3	S	45	50	0	5	45	100
KK2	M.M	0.90	1.00	27.72	56.08	20.0	E	85	15	0	0	0	40
L	H.S	1.07	1.12	26.68	54.25	18.8	N	72	17	8	3	0	40
S	H.S	0.23	0.36	28.03	44.75	20.3	S	45	44	5	5	1	120
WW1	M.M	0.35	0.72	27.44	52.33	21.0	NE	65	30	5	0	0	90
WW2	M.M	0.87	1.06	26.50	57.00	20.3	NW	75	25	0	0	0	30



(a) Coleford



(b) Goodhope 1



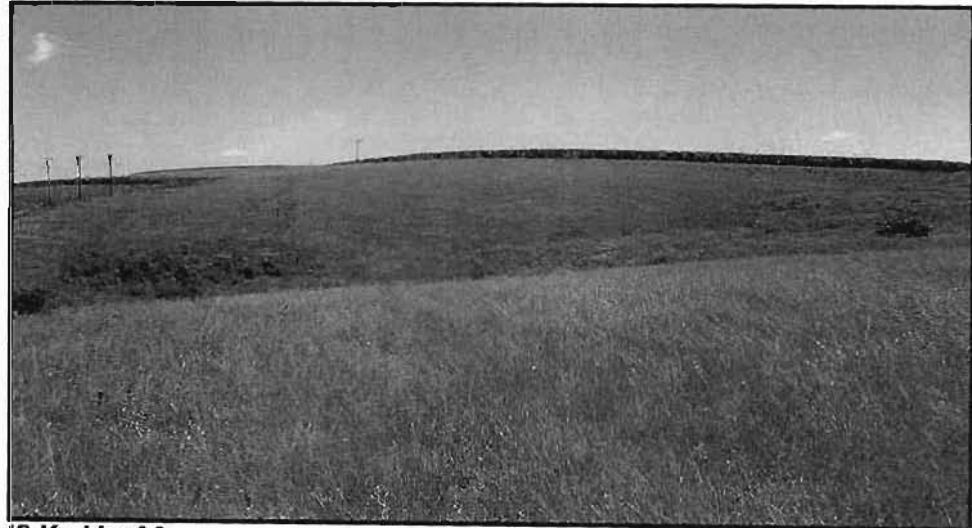
(c) Goodhope 2



(d) Himeville



(e) Karkloof 1



(f) Karkloof 2



(g) Linwood



(h) Stirling



(i) Wahrenonga 1



(j) Wahoonga 2

Figure 2 (a) to (j). Photographic representations of the ten study sites that were sampled over a twelve-month period in the KwaZulu-Natal Midlands.

2.2. Sampling

2.2.1. Grasses

Grasses were assessed at each site by taking 300 random points throughout the grassland. Grasses were only sampled once as the study sites were relatively stable entities. All grass species are perennial species and the management regime imparted stable, in some cases, it has not been altered for decades, resulting in a climax dominated species composition. Species turnover is thus limited due to the management and the stability of the grasslands sampled. Species turnover would only be noticeable in these conditions over an elongated time period, which falls outside the parameters of this study. Sampling was done in February 2001 when species were in flower for identification (Everson *et al.*, 1990). Grasses were identified using van Oudtshoorn (1992), Poulter (1993) and with support from Professor N.M. Tainton.

provided insight into seasonal information on grasshoppers and butterflies. This information is important, as it dictates the correct time of year to undertake sampling.

Transects were chosen randomly across the site, at each of the ten sites, ten, 50 m x 5 m transects were walked each month. Transects were marked out using a 50 m tape measure. No transects overlapped, with a minimum of 25 metres between transects to avoid counting individuals twice. Five metre wide transects were chosen as narrower width transects returned less data records, which may influence the statistical strength of the tests undertaken. Only adult grasshoppers were counted, as nymphs are difficult to identify to the species level. This is an accepted protocol and has been utilised by many authors (Samways, 1990; Samways & Moore, 1991; Armstrong and van Hensbergen, 1996; Chambers and Samways, 1998; Foord *et al.*, 2002; Gebeyehu and Samways, 2002). As the basis for a rapid assessment is reduced time expenditure, nymph identification is not feasible.

Orthoptera are inherently difficult to sample because of their jumping and flying behaviour (Browde *et al.*, 1992). Sampling techniques for assessing absolute densities are numerous and include 1) In situ or flush counts from a demarcated area 2) Night cages 3) Quick traps, and 4) Drop cages. The decreased mobility of grasshoppers at night (Anderson & Wright, 1952) undoubtedly contributes to the accuracy of night cage estimates. Sweep netting capture biases towards older grasshoppers. This left only one really suitable method. The orthopteran assemblage was visually assessed along each transect with supplementary sweep netting only being employed for the capture of unfamiliar species. Sweep netting was random with no predetermined pattern or number of sweeps per transect. This method has been well tried and shown to be the most effective one for sampling grasshoppers in South African grasslands (Samways, 1990; Samways & Moore, 1991; Prendini *et al.*, 1996; Chambers & Samways, 1998; Stewart, 1998; Gebeyehu and Samways 2002) and has been successfully utilised by (Kemp, 1992; Legg *et al.*, 1996; Onsager, 2000) in the United States of America and other areas (Sergeev, 1998; Zschokke *et al.*, 2000).

Voucher specimens were collected using a hand net. The specimens were pinned and dried and were identified by myself with support from Dr. Adrian Armstrong of the KwaZulu-Natal Nature Conservation Services. Dirsh (1965), Brown (1962) and Green

(1999) were used to aid identifications. Three species were not identified to species level due to their taxonomy being altered regularly as no definitive classification of the genera in question has been undertaken.

Data were collected between 09h00 and 14h00, only on days with no wind and no cloud cover. Before 09h00, it was still too cool and the grasshoppers were inactive. After 14h00, the wind increased, and the study area cooled down rapidly with reduced grasshopper activity.

All data for each of the sites was collected on the same day, in order to prevent sampling errors due to varying weather conditions.

2.2.3. Butterflies

Observations were made using the line transect method (Shapiro, 1975; Pollard, 1977, 1984; Thomas, 1983; Thomas & Mallorie, 1985; Holl, 1985; Swengel, 1990; Kremen, 1990; Pollard & Yates, 1993; Harding *et al.*, 1995). The length of transects were 50 metres. This method is now being extensively used to survey and monitor butterfly populations and assemblages (Shreeve & Mason, 1980; Erhardt, 1985; Warren *et al.*, 1986; Pollard and Yates, 1993; Kitahara & Sei, 2000). It is a method of significant value when differences in species abundances among sites are being investigated (Gall, 1985; New, 1991). Transect counts were done once a month for the length of the study period between 09h00 and 14h00 under fine weather conditions. Walking at a steady pace along the transect line (50 metre length), the number of adult individuals of each species sighted within a belt, ca. 5m wide, were recorded. Those individuals not immediately identified on the wing were captured by net and identified. Butterflies were identified by myself with support from Ms. Sarah Pryke, using the Natal Museum Collection and (Pringle *et al.*, 1994; Henning *et al.*, 1997).

2.2.4. Environmental variables

Ten environmental variables were measured to reflect the suspected predominant conditions of the grasslands for grasshoppers and butterflies: % Grass, % Forb, % Rock, % Bare ground and % Invasive vegetation were visually assessed, using random quadrat placements over the extent of the grassland (Table 3). Rock % and Bare

ground % were measured once off during the sampling of the grass species. The quadrats (25cm X 25cm) were placed randomly at 25 points across the site with the percentage of all the different environmental facets assigned a proportion within the quadrat boundary. Average vegetation height was measured using a tape measure. Vegetation height was measured each visit at 25 points across the site. When the grass sward was significantly altered, i.e. the site was burnt or mowed, these measurements were excluded from the average vegetation height calculation. The average vegetation of the site was representative of the average dominant management practice, namely, leaving a standing grass sward for grazing and water catchment protection. Relative Humidity (%) and Temperature ($^{\circ}\text{C}$) were measured using a hygrometer and thermometer respectively each month and were averaged. Elevation and Slope orientation were measured using a Global Positioning System (GPS, Garmin 2 plus) (Tables 1 and 3.).

Three other variables were measured, and were related to management practices: Burning (Burnt in 2001 or 2002), Grazing (Livestock grazed, Wildlife grazed, not grazed or Mown) (Table 2) and Veld Condition scores. The veld condition scores were computed for each of the ten grassland sites using Tainton and Camp (1999) assessment techniques, namely the 'key species' technique and the 'benchmark' technique. Both of these assessments involve the weighting of important species. In the 'key species' assessment ecologically important grasses (Tainton and Camp 1999), and in the 'benchmark assessment' agriculturally important species utilised by grazing livestock, are considered (Camp and Hardy 1999).

2.3. Statistical analysis

2.3.1. Data transformation

Grasshopper relative abundance values were highly skewed, with one species making up 50% of all the individuals recorded. To overcome the skewed distribution and normalise the data, records were log transformed. Due to the large number of zeros, the data were transformed using $Y = \log(X+1)$ (Zar, 1984). The data were log transformed throughout to maintain consistency in the results obtained. All data was Base 10 transformed. The feeding guild data were ARCSIN transformed, because they

were proportional data. The formula used in Excel to convert the proportions to workable values was:

$$\text{DEGREES (ASIN (SQRT (Proportion value/100)))}$$

2.3.2. Canonical ordination

Canonical ordination is a class of techniques used for relating species composition of the community to each of the environmental variables of the sample area (ter Braak & Šmilauer 1998). Correspondence Analyses (CA) were undertaken to express patterns in grasshopper assemblages. Canonical Correspondence Analyses (CCA) were undertaken to illustrate relationships between grasshopper species and each of the environmental variables (Kent & Coker 1992).

Colinearity of environmental variables occurred due to certain variables being strongly correlated to one another. Collinear environmental and nominal variables were omitted as one variable could explain their effects as a whole (ter Braak & Šmilauer 1998). An example of colinearity was '%Grass' that could account for and explain the key and benchmark veld scores. The higher the percentage grass, the better the score and rank for the veld scores. The %Grass could also account for %Rock, %Bareground, %Forb and %Alien invasive because the amount of grass directly affected the amount of the other variables. These variables were therefore classified as dependent variables.

Colinearity also occurred among nominal and environmental variables, as the number of variables was higher than the number of active samples (ter Braak & Šmilauer 1998). Due to colinearity having occurred because of a higher number of environmental variables than active samples, the nominal and environmental variables were separated and run as two individual groups.

The significance of the relationship between species and environmental variables was evaluated using Monte Carlo permutations. A Monte Carlo permutation tests statistical significance by repeated shuffling of the samples (ter Braak & Šmilauer 1998). A F-

value is worked out for the 199 permutations run under a reduced fit model and this value is then compared to the F-value for the original data set to prove significance.

Due to the large number of species encountered over the sampling period, only species with 50% or greater of their variance explained were plotted so as to reduce the number of species on any one plot.

A unimodal method of statistical analysis was used as a large number of zeroes contributed to the species matrix. The absence and species turnover across the sites and months was high and does not support the use of linear methods of data analysis.

Having used a unimodal method, rare species were having an unduly large influence on the species data analyses. The rare species were down weighted to reduce their emphasis on the outcome of the Canonical Correspondence Analysis (CCA).

In the Partial Correspondence Analysis (PCA) testing a numerical value was ascribed to each of the slope orientations recorded during the study period in order to be able to run the appropriate statistical tests.

The program used to do all the Correspondence Analysis (CA) Canonical Correspondence Analysis (CCA) and Partial Correspondence Analysis (PCA) was CANOCO 4 for windows.

2.3.3. Cluster Analysis and evenness indices

Cluster analysis was used to describe a set of numerical techniques in which the main purpose was to divide the objects of study into discrete groups or categories. These groups were based on the characteristics of the grasshoppers and its clusters have a significant role to play in linking similar sites and months. The unweighted, Bray-Curtis method of comparison was used as all the sites were sampled equally. The advantage of using the Bray-Curtis coefficient of similarity (Magurran, 1988) is that it is a quantitative measure that takes into account species abundances (Southwood, 1978; Magurran, 1988). The program used to undertake the cluster analysis was PC-ORD.

2.3.4. Analysis of Variance (ANOVA)

An analysis of variance (ANOVA) was done on the species composition between the different grazing regimes, feeding guilds and grassland types to measure any statistically significant difference between them. The level of significance was 95% $P = 0.05$. ANOVA's were done using STATISTICA 4 package and were transformed to comply with the assumptions that are made when doing an ANOVA (Johnson & Wichern, 1992).

2.3.5. Pearson Correlations

The general purpose of a Pearson Correlation is to learn more about the relationship between several independent variables and how congruent they are when comparing them across a number of samples. The major conceptual limitation of all correlation techniques is that one can only ascertain relationships, but never be sure about underlying causal mechanisms. The computer program used to do Pearson Correlations was STATISTICA 4.

2.3.6. Mantels Test

The Mantel test is used to test the null hypothesis of no relationship between two square symmetrical matrices. This test evaluates the correlation between distance (or similarity or dissimilarity) matrices (McCune & Grace, 2002). The distance measure used for the two matrices is the Sorenson measure (Bray-Curtis). The Mantel test is an alternative to regressing one correlation matrix against another, circumventing the problem of partial dependence within each matrix. Because the cells of distance matrices are not independent of each other, we cannot accept the p-values from standard techniques that assume independence (for example, Pearson correlation). Nevertheless, the Pearson correlation r can be used as a measure of the strength of the relationship between the two distance matrices. In this case, r is called the standardized Mantel statistic and ranges from -1 to 1 (Rohlf & Sokal, 1995).

1000 Monte Carlo simulations are under taken. These simulations randomize the data, by shuffling the order of the rows and columns. After each permutation is successfully completed, Mantel's Z-statistic is calculated with the resulting Z statistics from all

permutations providing an empirical distribution that is used for the significance test. PC-ORD was used to undertake the necessary simulations and testing.

2.3.7. Multiple Response Permutation Procedures (MRPP)

MRPP is a non-parametric procedure for testing the hypothesis of no difference between two classes or entities. Three assumptions need consideration when using MRPP.

1. The distance measure chosen adequately represents the variation of interest in the data.
2. The sample units are independent. The usual problems with pseudoreplication, subsampling and repeated measures are conceptually the same as with ANOVA.
3. The relative weighting of the variables has been controlled prior to calculating the distance measure, such that the weighting of variables is appropriate for assessing the ecological question being asked.

A natural weighting methodology recommended by Mielke (1984) was utilised, and is the most common weighting measure in recent applications of MRPP, being one of four offered by PC-ORD. Groups were separated across nine distinct categories. The categories are: ASPECT, FORB CONTENT, VEGETATION HEIGHT, GRASSLAND TYPE, VELD CONDITION, GRAZING, BIENNIAL BURN, DISTURBANCE and SEASON OF BURN. A Sorenson (Bray-Curtis) distance measure was utilised, as it is more robust and ecologically meaningful measure, as the data set that it tested had a high number of zeroes (absence data). The traditional Euclidean distance measure is less robust and usually only shows more linear discrepancies. The computer program utilised to elucidate differences in species assemblages between two measured classes was PC-ORD, Version 4.25.

When interpreting results, the test statistic T , describes the separation between the groups. The more negative is T , the stronger the separation between the two classes being compared, and therefore the more statistically significant the p -value.

2.3.8. Indicator Species Analysis (ISA)

This method of assessment combines information on the concentration of species abundance in a particular group and the faithfulness of the occurrence of that species to a particular group (Dufrene & Legendre, 1997). A perfect indicator is a species which is always present within a certain group (category) i.e. it is 'faithful' to that group, in addition it should also be 'exclusive' to that group, never occurring in other groups (McCune & Grace, 2002).

To achieve a result of whether a species is an indicator or not, the proportional abundance, and frequency of occurrence of each species is calculated for a particular group. Both values are then represented as a percentage and combined to produce a resultant value. The highest indicator value is 100 and the lowest is zero. A good indicator will have a high percentage for both proportional frequency and abundance; conversely a poor indicator will have low percentages for the two categories calculated. To validate the statistical significance of the values, they are randomly reassigned to different sampling units 1000 times using the Monte Carlo method (McCune & Grace, 2002). The program utilised to undertake ISA was PC-ORD, Version 4.25.

CHAPTER 3

RESULTS

3.1. Grass species composition and abundance

Thirty-two species of grass belonging to seven tribes were sampled during the February 2001 survey. One species was an exotic species, namely *Paspalum urvillei*. Only two species of grass were encountered across all ten sites (*Eragrostis curvula* and *Themeda triandra*) (Table 4). Six species of grass were encountered at one site only; *Brachiaria serrata*, *Digitaria eriantha*, *Eulalia villosa*, *Imperata cylindrica*, *Melinis nerviglumis* and *Miscanthus capensis*. Only two of these species, namely *B. serrata* and *M. capensis* were recorded in abundance, the remainder being rare (recorded ten times or less). In addition, six species were recorded at more than one site, but were rare (Table 4).

The most common grass was *T. triandra* making up greater than 50% of the grass composition at five of the ten sites (Coleford, Goodhope 2, Karkloof 2, Linwood and Wahroonga 2) (Table 4), which were predominantly short, open grasslands with an average vegetation height of between 30cm and 45cm. The average grass cover was between 75% and 85% (Table 3).

Wahroonga 1 had one grass species (*Hyparrhenia hirta*) making up greater than 50% of all grasses recorded. Stirling had 45% *Miscanthus capensis* and Karkloof 1, 41% *Aristida junciformis*. *A. junciformis* is a pioneer and invader species, hence its high abundance at Karkloof 1, a rehabilitated pine compartment.

Two sites Himeville and Goodhope 1 did not show similar compositional trends to the other grasslands with 50% or more of the grass composition at

these sites shared across three species. At Himeville (*Alloperis semialata*, *Heteropogon contortus* and *Tristachya leucothrix*) and Goodhope 1 (*Cymbopogon plurinodis*, *Eragrostis curvula* and *Themeda triandra*) (Table 4).

Table 4. Grass species abundances recorded during February 2001 across the ten study sites sampled. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)

Species	C	GH1	GH2	H	KK1	KK2	L	S	WW1	WW2
ANDROPOGONEAE										
<i>Cymbopogon excavatus</i> (Hochst.)								60		
<i>Cymbopogon plurinodis</i> (Stapf)		51	30						10	
<i>Diheteropogon amplexans</i> (Nees)		29	5		6				9	
<i>Eulalia villosa</i> (Thun.)	4									
<i>Heteropogon contortus</i> (L.)	21	3	3	93	1	2			2	44
<i>Hyparrhenia hirta</i> (L.)		23	42						162	9
<i>Imperata cylindrica</i> (L.)		1								
<i>Miscanthus capensis</i> (Nees)								135		
<i>Monocymbium cerissiforme</i> Stapf			28				13			58
<i>Themeda triandra</i> Forssk.	178	58	159	35	25	172	215	39	42	151
<i>Trachypogon spicatus</i> Kuntze				2		5				
ARUNDINELLEAE										
<i>Loudetia simplex</i> (Nees)						7			3	
<i>Tristachya leucothrix</i> Nees	86	15	2	65	12	7			20	
CHLORIDAE										
<i>Harpochloa falx</i> Kuntze					7	4				
<i>Rendlia altera</i> (Rendle)			2	9						
ERAGROSTEAE										
<i>Eragrostis capensis</i> (Thun.)	1			1					8	
<i>Eragrostis curvula</i> (Schrad.)	1	73	7	6	70	3	30	15	6	14
<i>Eragrostis plana</i> Nees			3							
<i>Eragrostis racemosa</i> (Thun.)		5	9		5	22	30			21
PANICEAE										
<i>Alloteropsis semialata eckloniana</i> Nees	2			86	18	4				
<i>Brachiaria serrata</i> (Thun.)						47				
<i>Digitaria eriantha</i> Steud.						1				
<i>Melinis nerviglumis</i> (Franch)					23					
<i>Melinis repens</i> (Willd)			4						2	
<i>Panicum ecklonii</i> Nees				1						
<i>Panicum natalense</i> Hochst.	7	5	3			1			6	
<i>Paspalum scrobiculatum</i> L.					4					
<i>Paspalum urvillei</i> Steud.		8						30		
<i>Setaria sphacelata sphacelata</i> (Schumach.)	14			1	6	20	1	6		
SPOROBOLAE										
<i>Sporobolus africanus</i> Robyns & Tournay			3				2			
<i>Sporobolus pyramidalis</i> P. Beauv.		12					1			
STIPEAE										
<i>Aristida junciformis</i> Trin. & Rupr.				1	123	5	8	15	30	3
Species richness	8	14	14	11	12	14	8	7	12	7

3.2. Veld condition scores

Coleford had the highest key and benchmark veld condition scores (1.35 and 1.26), resulting from a high percentage (88%) of the grass composition comprising *Themeda triandra* and *Tristachya leucothrix* (Table 3). These high scores reflect the high agricultural productivity of these grasslands. Five sites (Linwood, Karkloof2, Wahroonga 2, Goodhope 2 and Himeville) recorded scores over 0.8, these scores reflecting good condition grasslands (Table 3). The remaining four sites scored 0.56 or lower with Stirling being the poorest with a score of 0.23 and 0.36. This site is dominated by *Miscanthus capensis* that is not considered a key grass species in either the environmental or agricultural assessment (Camp and Hardy, 1998) (Table 3).

3.3. Forbaceous species composition and abundance

As this study was not aimed at determining the floral richness of sites, only species that were relatively common at any specific site or across sites were identified (Table 5). The site with the highest species richness was Wahroonga 1, a triennially burnt grassland, with limited wildlife utilisation. Three species encountered at this site are bulbous, and are typically excluded at sites, which are regularly burnt, or intensively grazed. The second most species rich site was Stirling, an over-utilised south facing grassland that had a profusion of woody plant species such as, *Leonotus leonurus*, *Phymaspermum acerosum* and *Rubus cuneifolius*. Two of the species encountered at this site were alien invasives, namely, *R. cuneifolius* and *Verbena bonariensis*. *P. acerosum* is considered an invasive plant in the midlands and usually becomes a problem in over-utilised lands or previously disturbed lands. *Senecio madagascarensis* is also indicative of disturbance and over-utilisation. The majority of the species found at Stirling were considered undesirable species, as they are invasive. The three most species poor sites are Himeville (3) and Coleford (2). Two of the most common species recorded at Himeville are alien species, with only one species being desirable. At Coleford only two species were recorded. The reason for the low plant species diversity is as a result of the farming practice imparted on the land prior to it being declared a nature reserve in 1947. This area when commercially farmed was regularly sprayed with broad-leaved weed-killers to improve the quantity of grass for grazing.

Table 5. A list of the most commonly encountered forbaceous plants and authorities (Germishuizen & Meyer, 2003) recorded during the study period at the ten study sites

Scientific Name	C	GH1	GH2	H	KK1	KK2	L	S	WW1	WW2
<i>Acalypha peduncularis</i> E.mey. Ex Meisn				*		*	*			
<i>Watsonia densiflora</i> Baker						*	*		*	
<i>Leonotus leonurus</i> (L.)					*			*	*	
<i>Hypoxis acuminata</i> Baker	*		*					*		*
<i>Hypoxis angustifolia</i> Lam.			*			*				*
<i>Apodolirion buchananii</i> Baker		*	*							
<i>Phymaspermum acerosum</i> (DC.) Kallersjo					*			*		
<i>Rubus cuneifolius</i> Pursh.				*	*			*		
<i>Verbena bonariensis</i> L.				*						
<i>Helichrysum aureonitens</i> Sch. Bip.	*		*					*		*
<i>Scilla natalensis</i> Planch.									*	
<i>Scilla nervosa</i> (Burch.) Jessop									*	
<i>Ledebouria revoluta</i> (L.f.) Jessop									*	
<i>Indigofera woodii</i> Bolus.						*			*	
<i>Felicia muricata</i> (Thunb.) Nees		*	*							*
<i>Helichrysum pilosellum</i> (L.f.) Harv.		*				*		*	*	*
<i>Senecio madagascarensis</i> Poir.		*						*		
<i>Berkheya rhapontica</i> (DC.) Huth. & Burt Davy								*		*
<i>Tephrosia polystachya</i> E. Mey.					*				*	
Species richness	2	4	5	3	4	5	3	7	8	6

3.4. Abundance across species

3.4.1. Grasshoppers

Table 6. Grasshopper species recorded during the study period, the code used in analyses, frequency of encounters for each species, the mean of species abundance, the mean per transect and the maximum number of individuals encountered at any one transect

Species	Species code	Abundance	Freq	Mean	L Mean	Max
ACRIDIDAE						
<i>Acorypha ferrifer</i> (Walker)	ACOFER	6	5	1.20		2
<i>Acorypha nigrovariegata tibialis</i> (Kirby)	ACOTIB	88	46	0.07	1.91	7
<i>Acrida</i> sp.	ACRUDW	1	1	0.00	1.00	1
<i>Acrida acuminata</i> (Stål.)	ACRACC	461	165	0.38	2.79	14
<i>Acrida bicolor</i> (Thunberg)	ACRBIC	25	23	0.02	1.09	2
<i>Anablepia dregei</i> (Ramme)	ANADRE	73	22	0.06	3.32	11
<i>Aneuryphymus montanus</i> (Brown)	ANEMON	10	8	0.01	1.25	3
<i>Anthermus granosus</i> (Stål.)	ANTGRA	77	42	0.06	1.83	7
<i>Calliptamulus natalensis</i> (Sjöstedt)	CALNAT	5	4	0.00	1.25	2
<i>Calliptimicus sermireseus</i> (Serville)	CALSEM	160	61	0.13	2.62	11
<i>Cannula gracilis</i> (Burmeister)	CANGRA	16	12	0.01	1.33	3
<i>Catantops melanostictus</i> (Schaum)	CATMEL	104	42	0.09	2.48	8
<i>Coryphosima stenoptera</i> (Schaum)	CORSTE	547	173	0.46	3.16	15
<i>Crucinotacris cruciata</i> (l. Bolivar)	CRUCRU	8889	607	7.41	14.64	94
<i>Cyrtacanthacris aeruginosa</i> (Stoll.)	CYRAER	38	25	0.03	1.52	3
<i>Dirshia abbreviata</i> (Brown)	DIRABB	98	46	0.08	2.13	9
<i>Eupropracris cylindricollis</i> (Schaum)	EUPCYL	2	2	0.00	1.00	1
<i>Eyprepocnemis plorans</i> (Charpentier)	EYPPLO	103	65	0.09	1.58	4
<i>Faureia milanjica</i> (Karsch)	FAUMIL	258	62	0.22	4.16	13
<i>Faureia rosea</i> (Uvarov)	FAUROS	6	5	0.01	1.20	2
<i>Gastrimargus crassicollis</i> (Saussure)	GASCRA	136	82	0.11	1.66	7
<i>Gastrimargus determinatus vitripennis</i> (Walker)	GASVIT	11	9	0.01	1.22	2
<i>Gastrimargus drakensbergensis</i> (Ritchie)	GASDRA	332	108	0.28	3.07	16
<i>Gymnobothrus temporalis temporalis</i> (Stål.)	GYMTEM	79	33	0.07	2.39	11
<i>Heteracris herbacea</i> (Serville)	HETHER	62	36	0.05	1.72	4
<i>Heteropternis guttifer</i> (Kirby)	HETGUT	6	5	0.01	1.20	2
<i>Macheridia bilineata</i> (Stål.)	MACBIL	294	158	0.25	1.86	5
<i>Omithacris cyanæ</i> (Stoll.)	ORNCYA	11	10	0.01	1.10	2
<i>Orthochtha rosacea</i> (Walker)	ORTROS	271	78	0.23	3.47	13
<i>Orthochtha zuluensis</i> (Popov and Fishpool)	ORTZUL	15	12	0.01	1.25	3
<i>Orthochtha dasycnemis</i> (Gerstaecker)	ORTDAS	2496	364	2.08	6.86	69
<i>Oxya hyla hyla</i> (Serville)	OXYHYL	134	13	0.11	10.31	39
<i>Paracinema tricolor</i> (Thunberg)	PARTRI	12	8	0.01	1.50	3
<i>Parga xanthoptera</i> (Stål.)	PARXAN	36	26	0.03	1.38	4
<i>Phaeocatantops sulphurius</i> (Walker)	PHASUL	33	29	0.03	1.14	2
<i>Pnorisa squalus</i> (Stål.)	PNOSQU	88	22	0.07	4.00	9
<i>Rhapholitta cephalica</i> (l. Bolivar)	RHACEP	62	17	0.05	3.65	12
<i>Scintharista rosacea</i> (Kirby)	SCIROS	3	3	0.00	1.00	1
<i>Vittacatantops botswana</i> (Jago)	VITBOT	350	159	0.29	2.20	14

LENTULIDAE						
<i>Eremidium basuto</i> (Brown)	EREBAS	144	59	0.12	2.44	10
<i>Lentula minuta</i> (Dirsh)	LENMIN	647	119	0.54	5.44	28
<i>Lentula obtusifrons</i> (Stål.)	LENOBT	302	130	0.25	2.32	12
<i>Qachasia fastigiata</i> (Dirsh)	QACFAS	148	64	0.12	2.31	10
PAMPHAGIDAE						
<i>Transvaaliana draconis</i> (Brown)	TRADRA	2	2	0.00	1.00	1
PYRGOMORPHIDAE						
<i>Dictyophorus spumans</i> (Thunberg)	DICSPU	23	13	0.02	1.77	4
<i>Maura rubroomata</i> (Stål.)	MAURUB	1	1	0.00	1.00	1
<i>Ochrophlebia caffra</i> (Linnaeus)	OCHCAF	91	55	0.08	1.65	5
<i>Phymateus leprosus</i> (Fabricius)	PHYLEP	2	2	0.00	1.00	1
TETRIGIDAE						
<i>Phloeonotus humilis</i> (Gerstaecker)	PHLHUM	109	56	0.09	1.95	10
<i>Phloeonotus</i> sp.	PHLSPP	6	4	0.01	1.50	2
<i>Tettiella</i> aff. <i>arcuata</i> (Hancock)	TETARC	12	8	0.01	1.50	3
TETTIGONIIDAE						
<i>Conocephalus</i> sp.	CONSPP	1346	255	1.12	5.28	30
Gryllidae sp.	GRYLLI	11	6	0.01	1.83	3
Tettigoniidae sp.	TETTIG	1	1	0.00	1.00	1
THERICLEIDAE						
<i>Whitea alticeps</i> (Descamps)	WHIALT	30	15	0.03	2.00	7

A total of 18273 individuals in 55 species, belonging to seven families and 18 sub-families, were recorded over a period from February 2001 to October 2002 at ten different sites (Table 6). The most abundant family was the Acrididae accounting for 71% of all grasshopper species. Two families recorded 7% respectively of the overall species composition, namely the Lentulidae and the Pygomorphidae. The remaining six families comprised 13% of the species assemblage. The most abundant species that occurred at all sites *Crucinotacris cruciata*, comprising 48.65% (8889 individuals) of the grasshopper observations (Table 6). The next most common species was *Orthochtha dasyncnemis* making up 13.66% (2496 individuals) of observations and was found at nine out of the ten sites sampled. *Conocephalus* sp. was the third most common species comprising 7.37% (1346 individuals) of the total observations. 17 species ranged from 3.54% (647 individuals) of observations to 0.56% (103 individuals) of observations while the remaining 35 species were 0.55% (100 individuals) of observations or lower (Table 6).

17 species of grasshopper recorded between 650 individuals and 100 individuals; *Lentula minuta* (647), *Coryphosima stenoptera* (547), *Acrida acuminata* (461), *Vittacatantops botswana* (350), *Gastrimargus drakensbergensis* (332), *Lentula*

obtusifrons (302), *Macheridia bilineata* (294), *Orthochtha rosacea* (271), *Faureia milanjica* (258), *Calliptimicus semiroseus* (160), *Qachasia fastigiata* (148), *Eremidium basuto* (144), *Gastrimargus crassicollis* (136), *Oxya hyla hyla* (134), *Phloeonotus humilis* (109), *Catantops melanostictus* (104) and *Eyprepocnemis plorans* (103).

15 species recorded less than 100 individuals; *Dirshia abbreviata* (98), *Ochrophlebia caffra* (91), *Pnorisa squalus* (88), *Acorypha nigrovariegata tibialis* (88), *Gymnobothrus temporalis temporalis* (79), *Anthermus granosus* (77), *Anablepia dregei* (73), *Rhaphotittha cephalica* (62), *Heteracris herbacea* (62), *Cyrtacanthacris aeruginosa* (38), *Parga xanthoptera* (36), *Phaeocatantops sulphurius* (33), *Whitea alticeps* (30), *Acrida bicolor* (25) and *Dictyophorus spumans* (23).

A further 14 species were encountered less than 20 times. *Scintharista rosacea*, *Calliptamulus natalensis*, *Acorypha ferriker*, *Faureia rosea*, *Heteropternis guttifera*, *Phloeonotus* sp., *Aneuryphymus montanus*, *Gryllidae* sp., *Gastrimargus determinatus vitripennis*, *Ornithacris cyanae*, *Paracinema tricolor*, *Tettiella* aff. *arcuata*, *Orthochtha zuluensis* and *Cannula gracilis*. All these species are classed as rare species for the sites that were sampled. Three species were recorded twice, *Transvaaliana draconis*, *Phymateus leprosus* and *Eupropracris cylindricollis*. Three species were recorded only once; they were *Acrida* sp., *Maura rubroomata* and a species of *Tettigoniidae* sp. (Table 6).

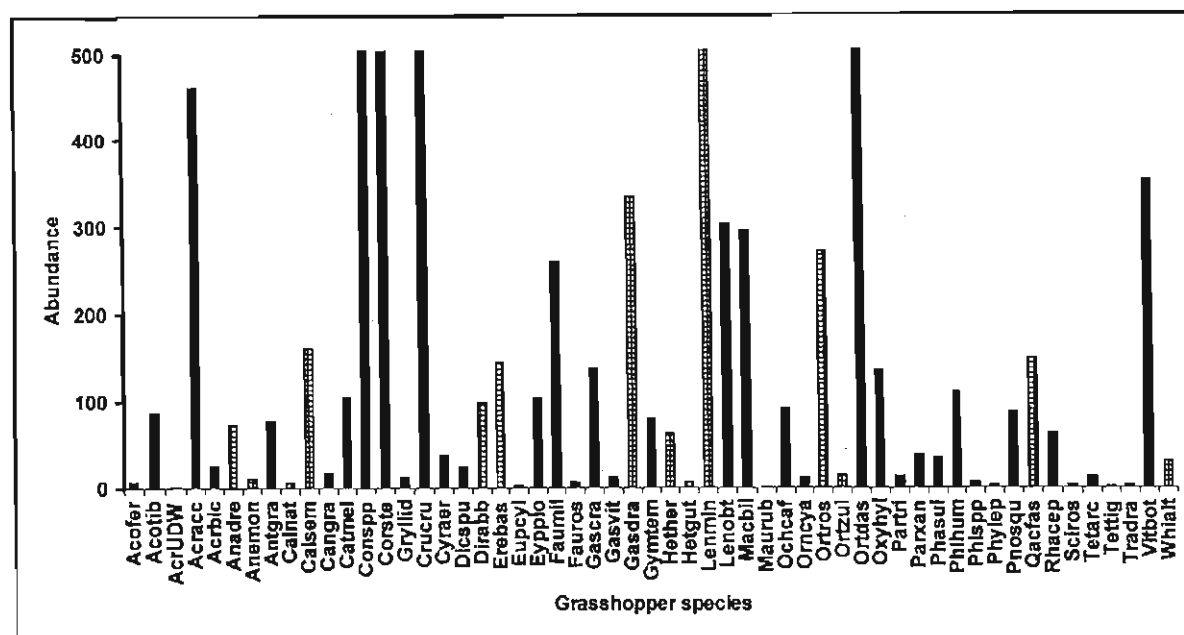


Figure 3. A histogram of grasshopper abundance over the 12-month study period for all 10 sites. (Species represented by hatched bars are endemic). Refer to Table 6 for grasshopper species codes.

3.4.2. Butterflies

Table 7. Butterfly species recorded during the study period. The code used in analysis, frequency of encounters for each species, the mean of species abundance, the mean per transect and the maximum number of individuals encountered at any one transect are given

Species	Species code	Abundance	Freq	Mean	L	Mean	Max
LYCAENIDAE							
<i>Actizera lucida</i> (Trimen)	ACTLUC	25	18	0.02	1.39	3	
<i>Aloeides oreas</i> (Tite & Dickson)	ALOORE	11	9	0.01	1.22	3	
<i>Aloeides taikosama</i> (Wallengren)	ALOTAI	9	6	0.01	1.5	3	
<i>Cupidopsis cissus</i> (Godart)	CUPCIS	14	10	0.01	1.4	3	
<i>Harpencyrtus noquasa</i> (Trimen & Bowker)	HARNOQ	6	6	0.01	1	1	
<i>Orachrysops ariadne</i> (Butler)	ORAARI	1	1	0	1	1	
<i>Orachrysops lacrimosa</i> (Bethune-Baker)	ORALAC	11	10	0.01	1.1	2	
<i>Orachrysops subravus</i> (Henning)	ORASUB	1	1	0	1	1	
NYMPHALIDAE							
<i>Acraea horta</i> (Linnaeus)	ACRHOR	2	2	0	1	1	
<i>Cassionympha cassius</i> (Godart)	CASCAS	77	46	0.06	1.67	8	
<i>Catacroptera cloanthe cloanthe</i> (Stoll.)	CATCLO	11	9	0.01	1.22	2	
<i>Danaus chrysippus</i> (Schreber)	DANCHR	78	64	0.07	1.22	3	
<i>Dingana bowkeri bowkeri</i> (Trimen)	DINBOW	15	10	0.01	1.5	3	
<i>Hyalites eponina eponina</i> (Cramer)	HYAEPO	2	2	0	1	1	
<i>Pardopsis punctatissima</i> (De Boisduval)	PARPUN	14	11	0.01	1.27	3	
<i>Precis hierta hierta</i> (Fabricius)	PREHIE	2	2	0	1	1	
<i>Precis octavia</i> Cramer)	PREOCT	69	46	0.06	1.43	5	
<i>Pseudonympha varii</i> (Van Son)	PSEVAR	64	40	0.06	1.68	5	
<i>Stygionympha wichgrafi wichgrafi</i> (Van Son)	STYWIC	19	3	0	1	1	
<i>Vanessa cardui</i> (Linnaeus)	VANCAR	72	49	0.07	1.8	7	

PAPILIONIDAE

<i>Papilio demodocus</i> (Esper)	PAPDEM	9	9	0.01	1	1
<i>Papilio nireus lyaeus</i> (Linnaeus)	PAPNIR	2	2	0	1	1

PIERIDAE

<i>Belenois aurota</i> (Fabricius)	BELAUR	35	25	0.03	1.4	3
<i>Belenois zochalia zochalia</i> (De Boisduval)	BELZOC	21	18	0.02	1.17	2
<i>Catopsilia florella</i> (Fabricius)	CATFLO	16	15	0.01	1.07	2
<i>Colias electo</i> (Fabricius)	COLELE	79	50	0.07	1.58	3
<i>Eurema brigitta brigitta</i> (Stoll.)	EURBRI	33	26	0.03	1.27	3

698 individuals, comprising 27 species and four families were recorded across the ten sites for the 12-month study period (Fig. 4). The most common butterfly was *Vanessa cardui* with 12.61% (88 individuals) of observations. Five other species were also abundant: *Colias electo* 11.31% (79), *Danaus chrysippus* 11.17% (78), *Cassionympha cassius* 11.03% (77), *Pseudonympha varii* 9.60% (67) and *Precis octavia* 9.55% (66). Four species were rare, only 20-35 individuals recorded: *Belenois aurota*, *Eurema brigitta brigitta*, *Actizera lucida* and *Belenois zochalia zochalia*. 17 species were very rare, recording less than 20 individuals, *Catopsilia florella* (16), *Dingana bowkeri bowkeri* (15), *Pardopsis punctatissima* (14), *Cupidopsis cissus* (14), *Orachrysops lacrimosa* (11), *Catacroptera cloanthe cloanthe* (11), *Aloeides oreas* (11), *Papilio demodocus* (9), *Aloeides taikosama* (9), *Harpencyreus noquasa* (6), *Stygionympha wichgrafi wichgrafi* (3), *Precis hierta hierta* (2), *Papilio nireus lyaeus* (2), *Hyalites eponina eponina* (2), *Acraea horta* (2), *Orachrysops subravus* (1) and *Orachrysops ariadne* (1) (Table 7).

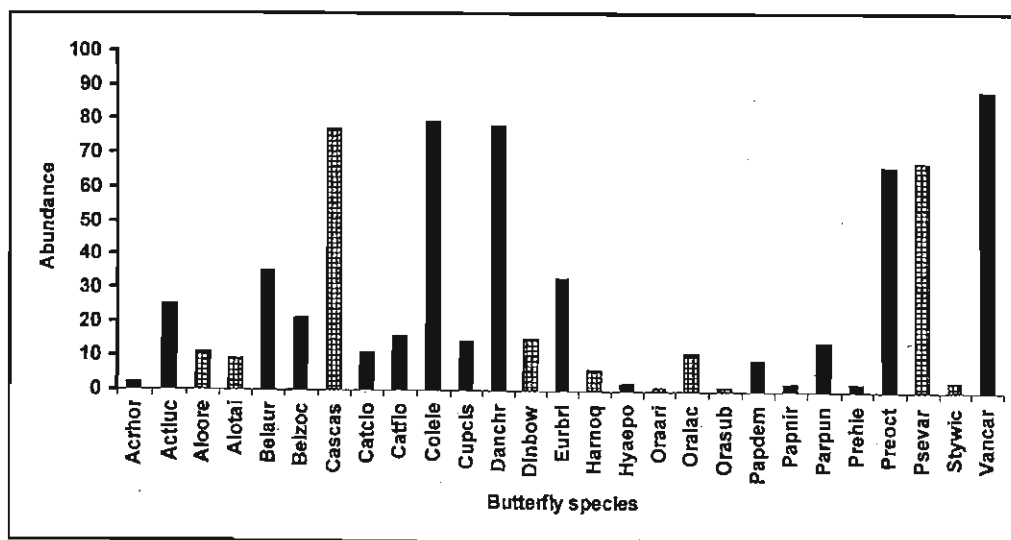


Figure 4. Butterfly species recorded across ten sites for the 12-month study period. (Species represented by hatched bars endemic). Refer to Table 7 for butterfly species codes.

3.5. Richness and abundance across sites

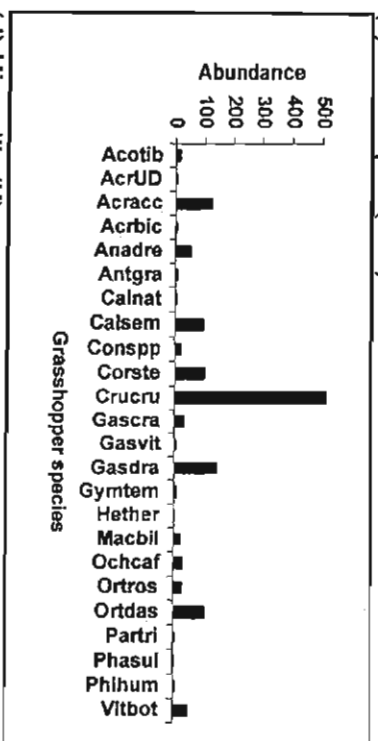
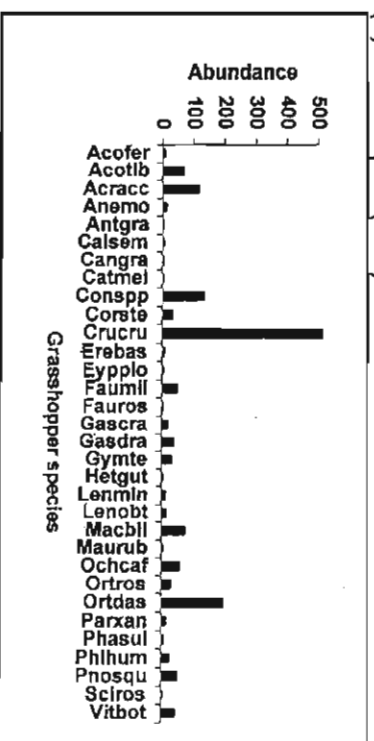
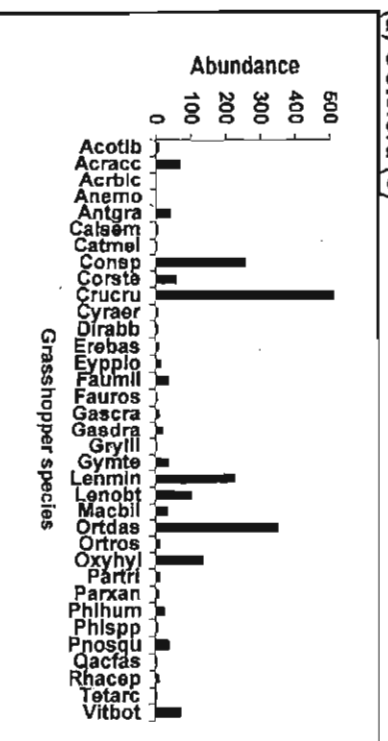
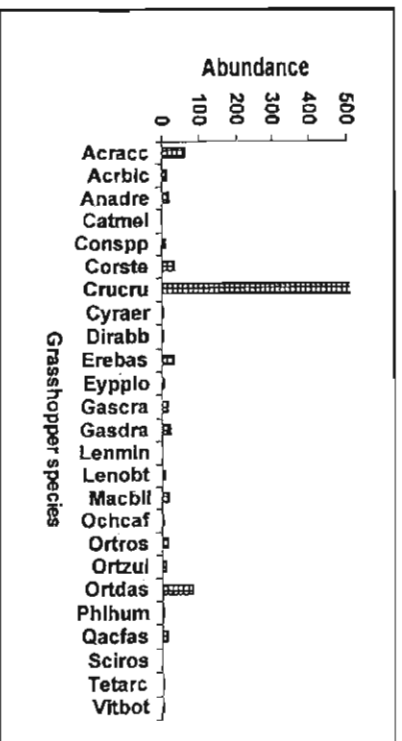
3.5.1. Grasshoppers

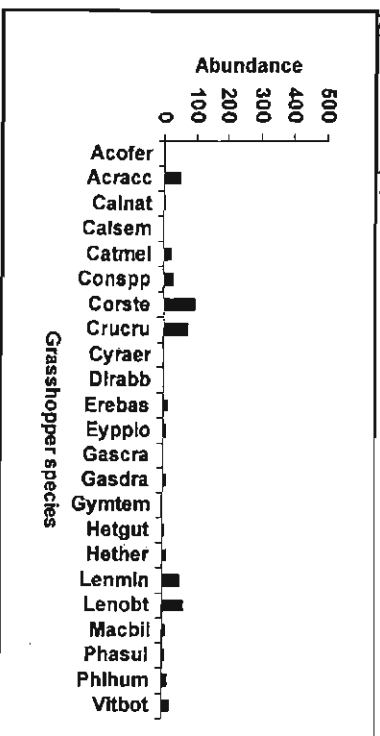
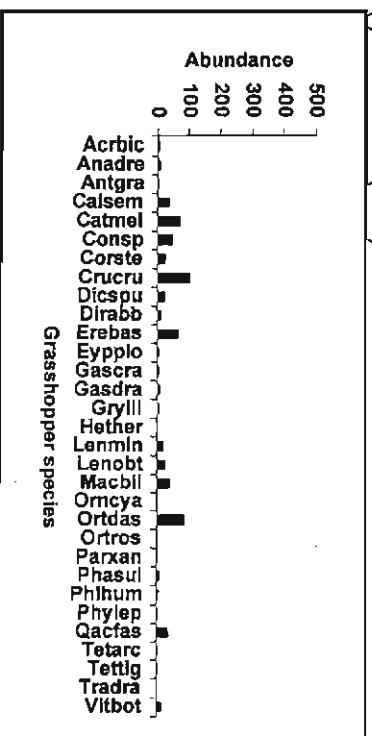
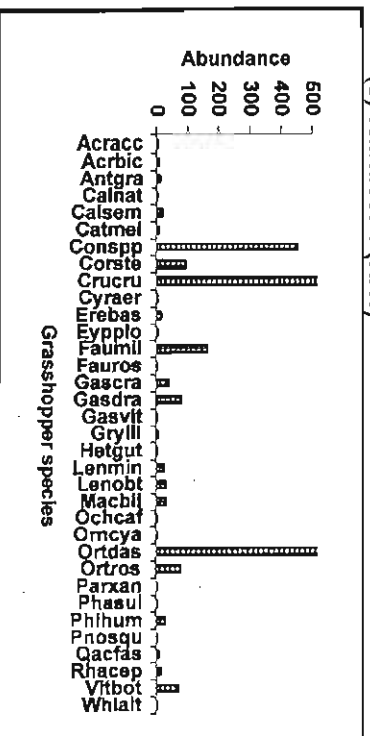
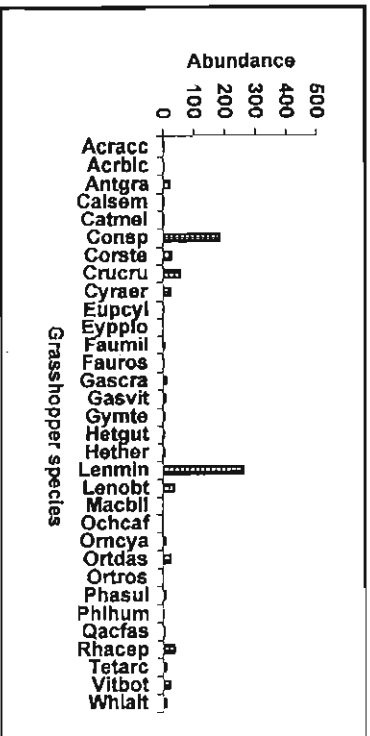
Of the 10 sites that were sampled between February 2001 and October 2002, the richest site was Goodhope 1 with 35 species (2397 individuals). Karkloof 2 was the second richest site (Table 8) having 34 species (3186 individuals). Karkloof 1 was represented with 32 species (755 individuals). The fourth richest site Goodhope 2 had 31 species (2145 individuals). Linwood had 30 species (619 individuals). The remaining five sites ranged from Wahrenonga 1 with 25 species (710 individuals) through Coleford 25 species (1800 individuals), Himeville and Stirling both had 23 species (3671 and 486 individuals) respectively. The most species-poor site was Wahrenonga 2 that only recorded 22 species (2504 individuals) (Fig. 5) (Table 8).

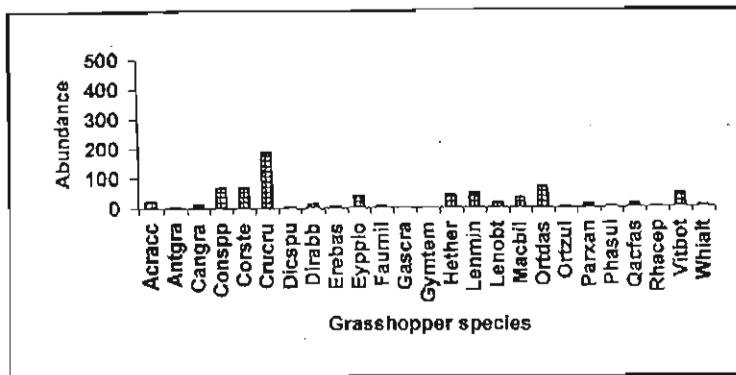
Table 8. Grasshopper diversity indices, evenness and species richness for each of the 10 sites between February 2001 and October 2002 (Data combined and log transformed). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2)

Site	Shannon's	Evenness (Pielou's J)	Abundance	Species richness
C	0.454	0.325	1800	25
GH1	1.024	0.663	2397	35
GH2	0.882	0.586	2145	32
H	0.484	0.350	3671	24
KK1	0.964	0.640	755	32
KK2	0.851	0.556	3186	34
L	1.170	0.785	619	31
S	1.087	0.798	486	23
WW1	1.102	0.789	710	25
WW2	0.642	0.478	2504	22

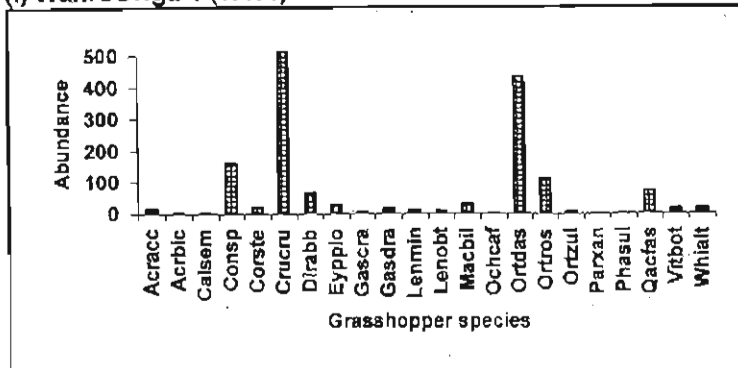
Evenness across sites was low. The most even site was Stirling (S) with a value of 0.798. The most uneven site was Coleford (C) with a value of 0.325. This site was dominated by *Crucinotacris cruciata* during the winter period. Stirling (S) was species poor, in comparison to Karkloof 1, 2 and Goodhope 1 that all had 32 or more species.







(i) Wahoonga 1 (WW1)



(j) Wahoonga 2 (WW2)

Figure 5 (a) to (j). Abundance of grasshopper species across each of the 10 study sites over the whole study period. Hatched bars represent Midlands Mistbelt and full bars Highland Sourveld. Refer to Table 6 for grasshopper species codes.

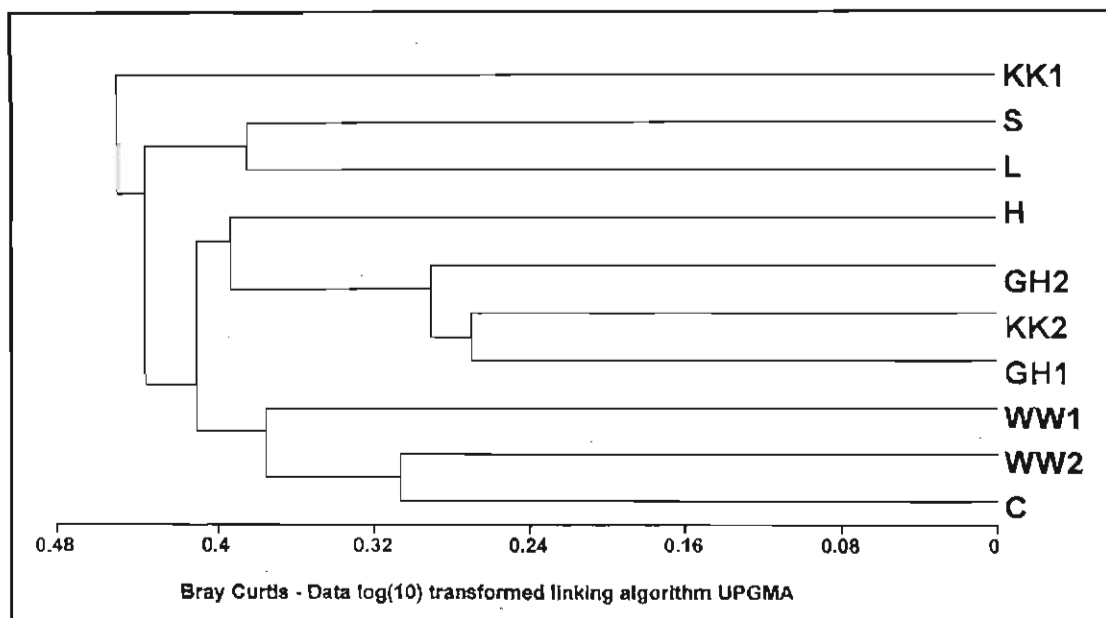


Figure 6. Grasshopper similarity across sites using Bray-Curtis Cluster Analysis. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahoonga 1, WW2= Wahoonga 2).

The Bray–Curtis Cluster Analysis (Fig. 6) linked together sites that showed similar species composition. It was interesting to note that the sites did not separate out according to the different grassland types sampled, i.e. Highland Sourveld and Midlands Mistbelt. There were four separate clusters of sites. The first cluster comprised of Karkloof 1 Midlands Mistbelt Grassland, a rehabilitated pine plantation, predominantly made up of *Phymaspermum acerosum*, a woody plant species standing about one metre high. Stirling and Linwood were both Highland Sourveld grasslands. Stirling was predominantly covered by tall vegetation, comprising woody species and *Miscanthus capensis* a tall grass species.

The third cluster comprised four sites, Himeville, Goodhope 1, 2 and Karkloof 2. Himeville and Goodhope 2 were both Highland Sourveld, the other two sites were Midlands Mistbelt grassland.

The fourth cluster comprised three sites, namely, Wahroonga 1, 2 and Coleford. Wahroonga 1 and 2 were expected to be similar as they were separated by a small grassland area. Both were Midlands Mistbelt grasslands, simply exposed to different management regimes.

Table 9. A presence-absence table of grasshopper species for the ten sites over the study period (* denotes presence). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)

Grasshopper Species	C	GH1	GH2	H	KK1	KK2	L	S	WW1	WW2
ACRIDIDAE										
<i>Acorypha ferrifer</i>			*					*		
<i>Acorypha nigrovariegata tibialis</i>		*	*	*						
<i>Acrida</i> sp.				*						
<i>Acrida acuminata</i>	*	*	*	*	*	*		*	*	*
<i>Acrida bicolor</i>	*	*		*	*	*	*			*
<i>Anablepia dregei</i>	*			*				*		
<i>Aneuryphymus montanus</i>		*	*							
<i>Anthermus granosus</i>		*	*	*	*	*	*		*	
<i>Calliptamulus natalensis</i>				*		*		*		
<i>Calliptimicus semiroseus</i>		*	*	*	*	*	*	*		*
<i>Cannula gracilis</i>				*					*	
<i>Catantops melanostictus</i>	*	*	*		*	*	*	*		
<i>Coryphosima stenoptera</i>	*	*	*	*	*	*	*	*	*	*
<i>Cruciotacris cruciata</i>	*	*	*	*	*	*	*	*	*	*
<i>Cyrtacanthacris aeruginosa</i>	*	*			*	*		*		
<i>Dirshia abbreviata</i>	*	*						*	*	*
<i>Eupropracris cylindricollis</i>					*					
<i>Eyprepocnemis plorans</i>	*	*	*		*	*	*	*	*	*

<i>Faureia milanjica</i>	*	*		*	*			*		
<i>Faureia rosea</i>	*	*		*	*					
<i>Gastrimargus crassicollis</i>	*	*	*	*	*	*	*	*	*	
<i>Gastrimargus determinatus vitripennis</i>				*	*	*				
<i>Gastrimargus drakensbergensis</i>	*	*	*	*		*	*	*	*	
<i>Gymnbothrus temporalis temporalis</i>		*	*	*	*			*	*	
<i>Heteracris herbacea</i>				*	*		*	*	*	
<i>Heteropternis guttifer</i>			*		*	*		*		
<i>Macheridia bilineata</i>	*	*	*	*	*	*	*	*	*	
<i>Ornithacris cyanae</i>					*	*	*			
<i>Orthochtha dasyncnemis</i>	*	*	*	*	*	*	*	*	*	
<i>Orthochtha rosacea</i>	*	*	*	*	*	*	*		*	
<i>Orthochtha zuluensis</i>	*							*	*	
<i>Oxya hyla hyla</i>		*								
<i>Paracinema tricolor</i>		*		*						
<i>Parga xanthoptera</i>		*	*			*	*	*	*	
<i>Phaeocatantops sulphurius</i>			*	*	*	*	*	*	*	
<i>Pnorisa squalus</i>		*	*			*				
<i>Rhaphotittha cephalica</i>		*			*	*		*		
<i>Scintharista rosacea</i>	*		*							
<i>Vittacatantops botswana</i>	*	*	*	*	*	*	*	*	*	
LENTULIDAE										
<i>Eremidium basuto</i>	*	*	*			*	*	*	*	
<i>Lentula minuta</i>	*	*	*		*	*	*	*	*	
<i>Lentula obtusifrons</i>	*	*	*		*	*	*	*	*	
<i>Qachasia fastigiata</i>	*	*			*	*	*	*	*	
PAMPHAGIDAE										
<i>Transvaaliana draconis</i>							*			
PYGOMORPHIDAE										
<i>Dictyophorus spumans</i>							*	*		
<i>Maura rubroornata</i>			*							
<i>Ochrophlebia caffra</i>	*		*	*	*	*			*	
<i>Phymateus leprosus</i>							*			
TETRIGIDAE										
<i>Phloeonotus humilis</i>	*	*	*	*	*	*	*	*		
<i>Phloeonotus sp.</i>		*								
<i>Tettiella aff. arcuata</i>	*	*			*		*			
TETTIGONIIDAE										
<i>Conocephalus sp.</i>	*	*	*	*	*	*	*	*	*	
Gryllidae sp.		*				*	*			
Tettigoniidae sp.							*			
THERICLEIDAE										
<i>Whitea alticeps</i>					*	*		*	*	
Species richness	25	35	32	24	32	34	31	23	25	22

Six species were habitat tolerant and recorded at all ten sites; *Conocephalus sp.*, *Coryphosima stenoptera*, *Crucinotacris cruciata*, *Gastrimargus crassicollis*, *Macheridia bilineata* and *Vittacatantops botswana* (Table 9).

Five species were recorded at nine of the ten sites; *Acrida acuminata*, *Eyrepocnemis plorans*, *Lentula minuta*, *Lentula obtusifrons* and *Orthochtha dasycnemis*. Two species occurred exclusively on Midlands Mistbelt grasslands, both species were at four of the five sites sampled, *Rhaphotittha cephalica* and *Whitea alticeps*. One species, *Anablepia dregei* occurred exclusively on Highland Sourveld grasslands (Table 9).

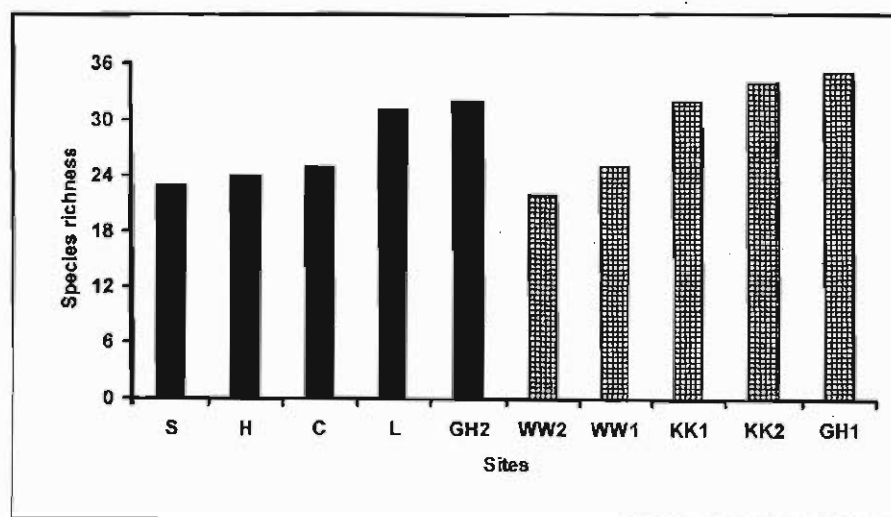


Figure 7. A bar chart of species richness for the ten sites sampled during the 12-month study period. (The hatched bars represent Highland Sourveld grasslands and the black bars Midlands Mistbelt). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2).

3.5.2. Butterflies

Table 10. Presence-absence of butterfly species for the ten sites over the whole study period (* denotes presence). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2)

Species	C	GH1	GH2	H	KK1	KK2	L	S	WW1	WW2
LYCAENIDAE										
<i>Actizera lucida</i> (Trimen)			*	*	*	*		*	*	*
<i>Aloeides oreas</i> (Tite & Dickson)	*	*	*							*
<i>Aloeides taikosama</i> (Wallengren)	*	*							*	*
<i>Cupidopsis cissus</i> (Godart)			*		*	*		*		*
<i>Harpentryreus noquasa</i> (Trimen & Bowker)		*	*		*					
<i>Orachrysops ariadne</i> (Butler)					*					
<i>Orachrysops lacrimosa</i> (Bethune-Baker)	*	*				*			*	*
<i>Orachrysops subravus</i> (Henning)								*		
NYMPHALIDAE										
<i>Acraea horta</i> (Linnaeus)					*					*
<i>Cassionympha cassius</i> (Godart)		*	*		*	*	*	*	*	*
<i>Catacroptera cloanthe cloanthe</i> (Stoll.)		*	*	*	*	*	*		*	*
<i>Danaus chrysippus</i> (Schreber)	*	*	*	*	*	*	*	*	*	*

<i>Dingana bowkeri bowkeri</i> (Trimen)						*			*	
<i>Hyalites eponina eponina</i> (Cramer)									*	
<i>Pardopsis punctatissima</i> (De Boisduval)				*	*			*		
<i>Precis hierta hierta</i> (Fabricius)								*	*	
<i>Precis octavia</i> Cramer	*	*	*	*	*	*	*	*	*	
<i>Pseudonympha varii</i> (Van Son)			*				*	*	*	
<i>Stygionympha wichgrafi wichgrafi</i> (Van Son)	*		*	*		*	*		*	
<i>Vanessa cardui</i> (Linnaeus)	*		*	*	*	*	*	*	*	
PAPILIONIDAE										
<i>Papilio demodocus</i> (Esper)		*						*	*	
<i>Papilio nireus lyaeus</i> (Linnaeus)			*					*		
PIERIDAE										
<i>Belenois aurota</i> (Fabricius)	*	*	*		*	*	*	*	*	
<i>Belenois zochalia zochalia</i> (De Boisduval)	*	*	*	*	*	*	*		*	
<i>Catopsilia florella</i> (Fabricius)	*	*		*		*	*	*	*	
<i>Colias electo</i> (Fabricius)	*	*	*	*	*	*	*	*	*	
<i>Eurema brigitta brigitta</i> (Stoll.)		*	*	*	*	*	*	*	*	
Species richness	11	14	16	10	15	15	9	15	16	22

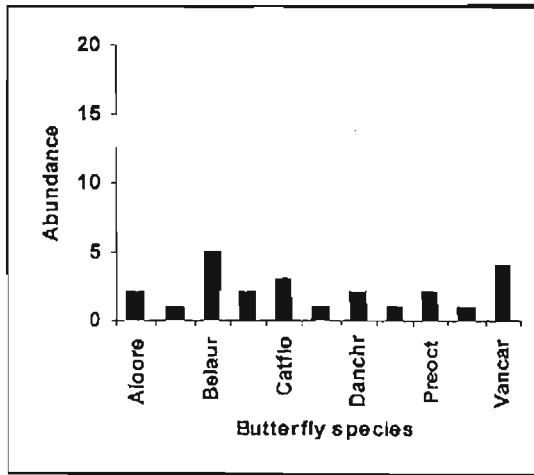
Table 11. Butterfly diversity indices, Shannon-Wiener, evenness and species richness for each of the ten sites over the whole study period (Data combined and log transformed). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2)

Site	Shannon's	Evenness (Pielou's J)	Abundance	Species richness
C	0.974	0.936	24	11
GH1	0.938	0.818	50	14
GH2	0.977	0.812	72	16
H	0.724	0.724	110	10
KK1	1.030	0.876	76	15
KK2	0.959	0.815	96	15
L	0.807	0.845	43	09
S	1.015	0.863	50	15
WW1	0.872	0.725	92	16
WW2	1.184	0.882	85	22

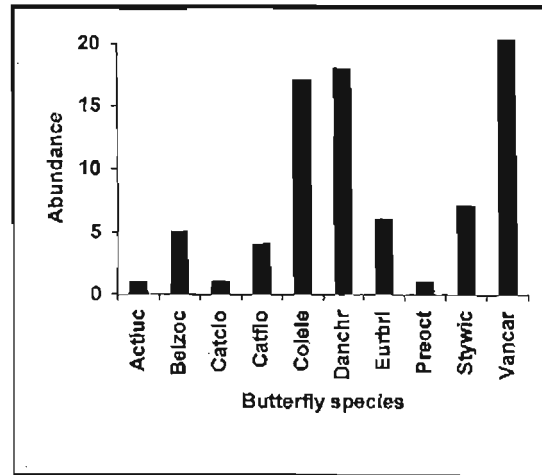
The evenness of the values for Pielou's J were similar signifying that individuals were distributed evenly across the species, with no one dominant species. Certain sites showed less evenness, namely, Himeville (H) and Wahrenonga 1 (WW1) with values of 0.724 and 0.725 respectively (Table 11).

Bar graphs (Fig. 8 (a) to (j)) represented the number of species and actual abundance for each of the ten sites over the whole study period. Of the ten sites that were sampled during this period the most species-rich site was Wahrenonga 2, recording 22 species (85 individuals) followed by Wahrenonga 1 and Goodhope 2 having recorded 16 species with 92 and 72 individuals respectively. Three sites, namely Karkloof 1, Karkloof 2 and Stirling recorded 15 species with 76, 96 and 50 individuals respectively.

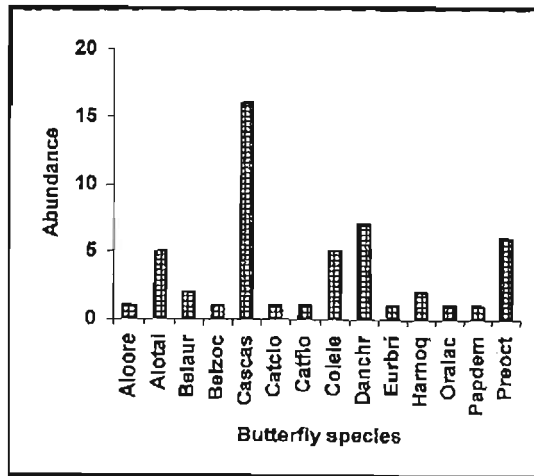
The four remaining sites recorded 14 species (50) (Goodhope 1), 11 species (24) (Coleford), 10 species (110) (Himeville) and nine species (43) for Linwood (Table 11).



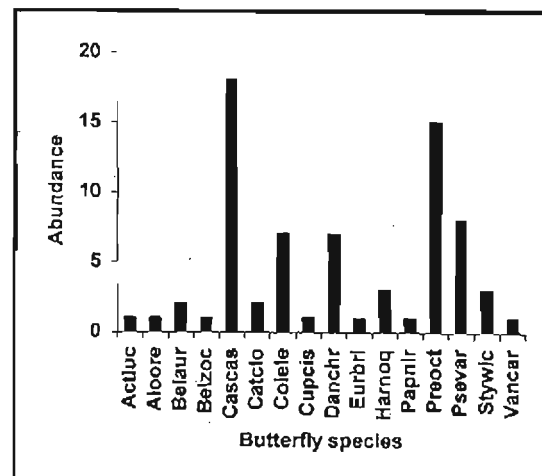
(a) Coleford



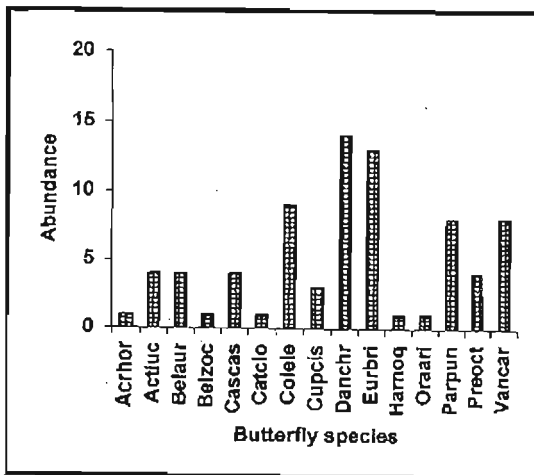
(b) Himeville



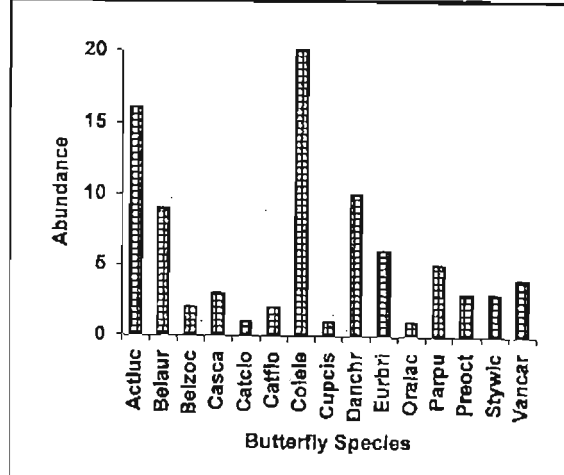
(c) Goodhope 1



(d) Goodhope 2



(e) Karkloof 1



(f) Karkloof 2

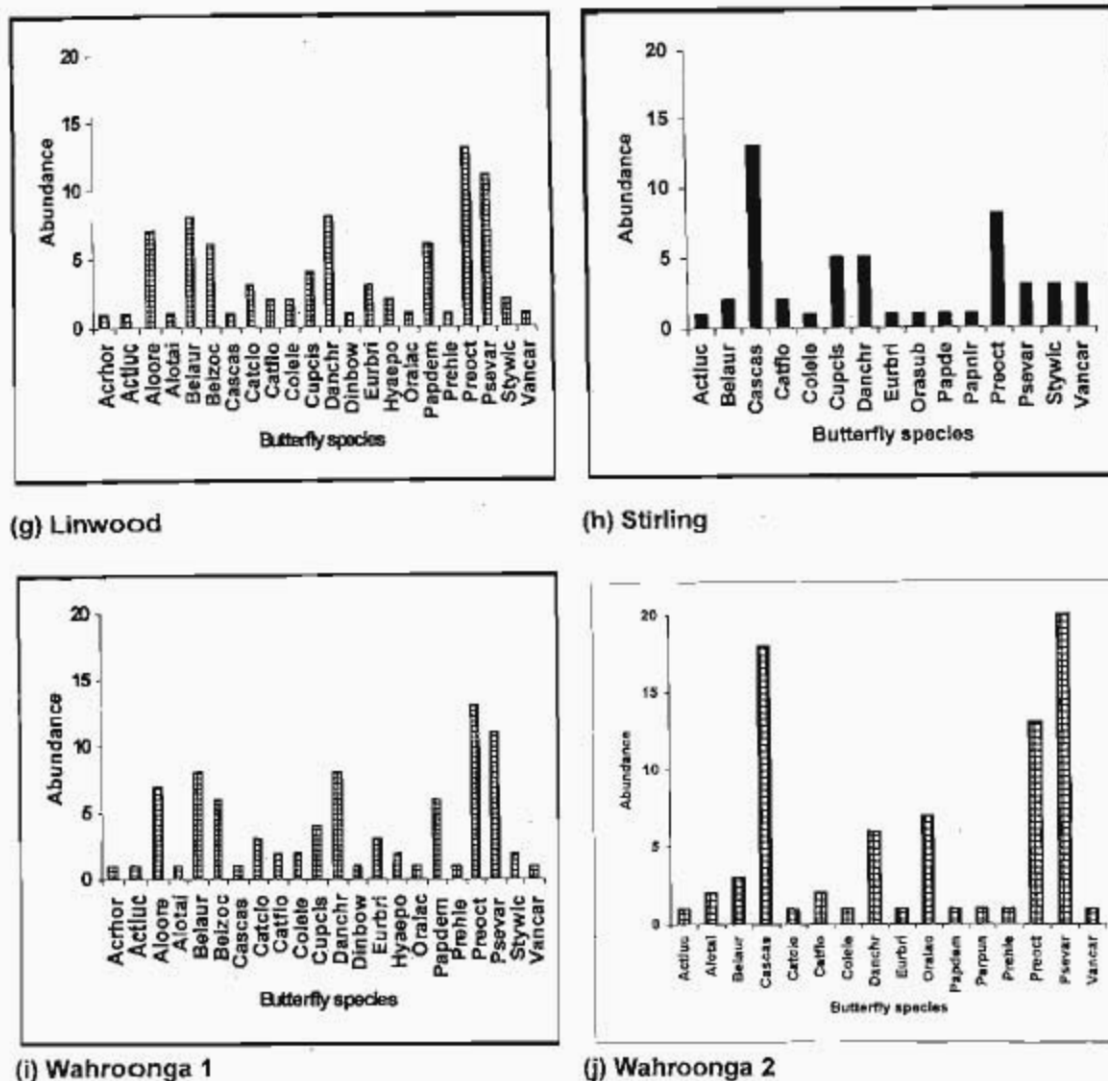


Figure 8 (a) to (j). Butterfly abundance per species for the whole study period. (Hatched bars represent Midlands Mistbelt grasslands, Black, Highland Sourveld). Refer to Table 7 for butterfly species codes.

A Bray Curtis Cluster Analysis (Fig. 9) was done to show the relationship between the ten sites according to butterfly species assemblages. Linwood and Coleford were the most dissimilar sites. Karkloof 1, 2 and Himeville were clustered closely, while Wahroonga 1, 2, Stirling, Goodhope 1 and 2 were similar to each other.

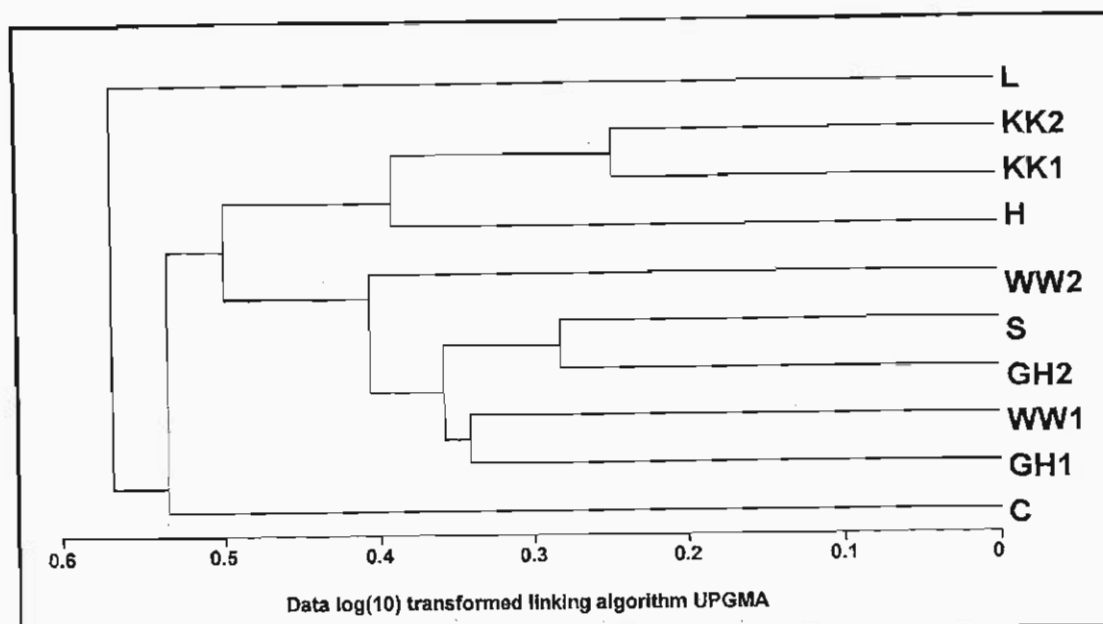


Figure 9. Butterfly similarity using Bray-Curtis Cluster Analysis across sites. (Data were combined and log transformed). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).

3.6. Endemic species

3.6.1. Grasshoppers

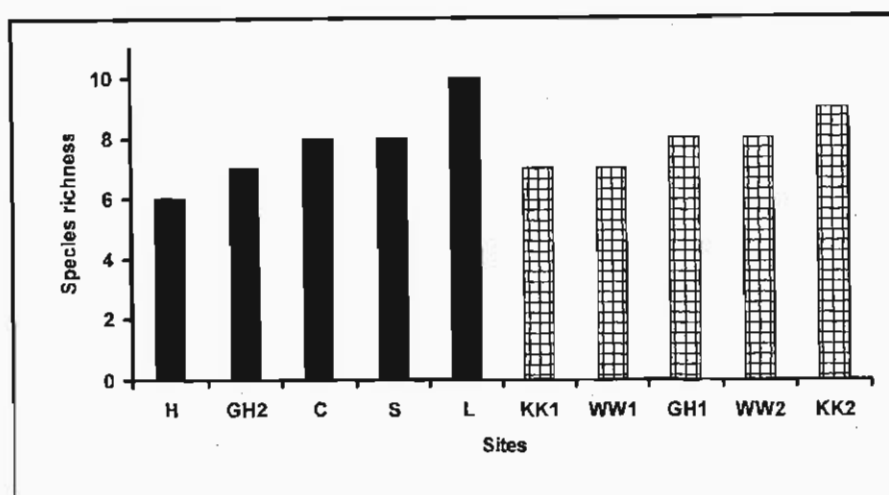


Figure 10. A bar chart of species endemism for the ten sites sampled during the 12-month study period. (The black bars represent Highland Sourveld grasslands and the hatched bars Midlands Mistbelt). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).

A total of 15 species were endemic to South Africa with five species montane endemics (Table 12). Endemicity did not vary significantly between Highland Sourveld and Midlands Mistbelt Grasslands, with 14 and 13 species respectively. Linwood had

the most endemic species, ten (Fig. 10). Linwood was Highland Sourveld grassland that was ungrazed and burnt biannually. Karkloof 2 had nine endemic species and was subject to the same management regime as Linwood. Himeville recorded six endemic species the lowest of all sites, yet it had the highest abundance of grasshoppers. Six endemic species were recorded in low abundance, between 2 and 30 individuals, three species recorded between 60 and 100 individuals and the six remaining species recorded between 140 and 647 individuals (Table 6).

Table 12. A presence absence table of endemic grasshopper species for the ten sites over the 12-month study period (* denotes presence). (Abundance measure, Low = (2-30), Medium = (60-100) and High = (140-647)). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2)

Species	Abundance	C	GH1	GH2	H	KK1	KK2	L	S	WW	WW
<i>Anablepia dregei</i>	Medium	*			*				*		
<i>Aneuryphymus montanus</i>	Low		*	*							
<i>Calliptamulus natalensis</i>	Low				*		*		*		
<i>Calliptimicus semiroseus</i>	High		*	*	*	*	*	*	*		*
<i>Dirshia abbreviata</i>	Medium	*	*					*	*	*	*
<i>Eremidium basuto</i>	High	*	*	*			*	*	*	*	
<i>Gastrimargus drakensbergensis</i>	High	*	*	*	*		*	*	*		*
<i>Heteracris herbacea</i>	Medium				*	*		*	*	*	
<i>Heteropternis guttifera</i>	Low			*		*	*		*		
<i>Lentula minuta</i>	Medium	*	*	*		*	*	*	*	*	*
<i>Orthochtha rosacea</i>	Low	*	*	*	*	*	*	*			*
<i>Orthochtha zuluensis</i>	Low	*								*	*
<i>Qachasia fastigiata</i>	High	*	*			*	*	*		*	*
<i>Transvaaliana draconis</i>	Low							*			
<i>Whitea alticeps</i>	Low					*	*			*	*
Species richness		8	8	7	6	7	9	10	8	7	8

3.6.2. Butterflies

Table 13. A presence absence table of endemic butterfly species for the ten sites over the 12-month study period (* denotes presence). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2)

Species	C	GH1	GH2	H	KK1	KK2	L	S	WW1	WW2
<i>Acraea horta</i>					*					*
<i>Aloeides oreas</i>	*	*	*							*
<i>Aloeides taikosama</i>	*	*							*	*
<i>Cassionympha cassius</i>		*	*		*	*	*	*	*	*
<i>Dingana bowkeri bowkeri</i>							*			*
<i>Harpenderus noquasa</i>		*	*		*					
<i>Orachrysops ariadne</i>					*					
<i>Orachrysops lacrimosa</i>	*	*				*			*	*
<i>Orachrysops subravus</i>								*		
<i>Pseudonympha varii</i>			*				*	*	*	*
<i>Stygionympha wichgrafi wichgrafi</i>	*		*	*		*		*	*	*
Species richness	4	5	5	1	4	3	3	4	4	8

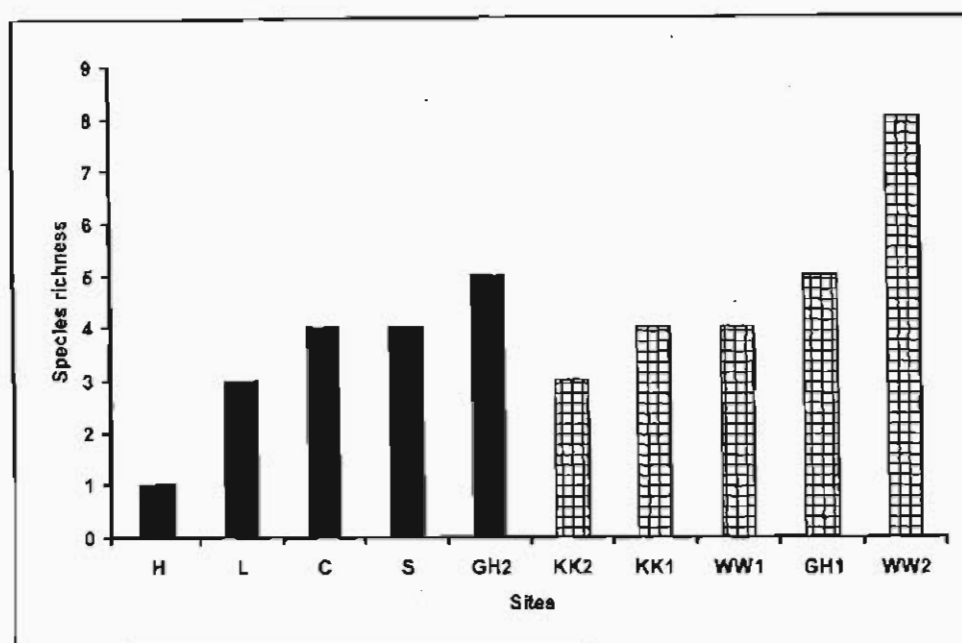


Figure 11. A bar chart of butterfly species endemism for the ten sites sampled during the whole study period. (The blue bars represent Highland Sourveld grasslands and the red bars Midlands Mistbelt). (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2).

There was no significant difference when comparing species richness across grassland type with Highland Sourveld (9 species) and Midlands Mistbelt (10 species). 11 species of butterfly were endemic to South Africa (Table 13). The most speciose site was

Wahroonga 2 (WW2) with eight endemics (72%) (Fig. 11). The second most speciose sites were the two Goodhope sites that recorded five endemics. The most species poor site was Himeville (H) that only recorded one endemic species of butterfly.

3.7. Seasonal variation

3.7.1. Grasshoppers

A Bray-Curtis Cluster Analysis (Fig. 12), that grouped together similar months in terms of species composition, agrees with the Correspondence Analysis, grouping together May through September and December through March. The cluster analysis also showed that the transitional months tended towards the 'summer' period rather than the 'winter' period in terms of their species composition. In the 'summer' period (December to March), the species richness ranged from 29 species (December) to 38 species (March) across all ten sites (Fig. 13, (a), (b), (c) and (l)). In the 'winter' period, species richness declined sharply, and ranged from six species (June) to 20 species (May) (Fig. 13, (e), (f), (g), (h) and (i)).

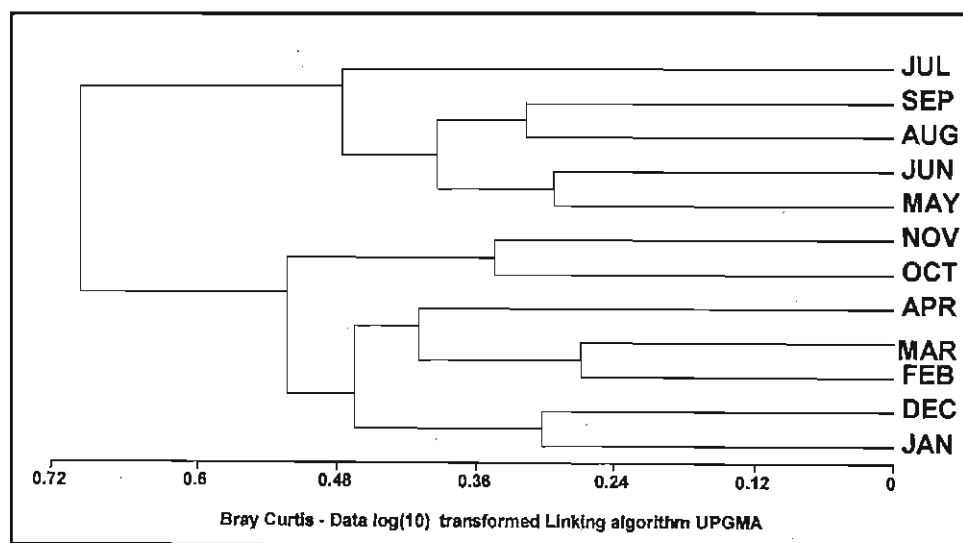


Figure 12. Grasshopper similarity across months using a Bray-Curtis Cluster Analysis for all sites (Combined site data).

Table 14. A presence absence table of grasshopper species during each month of the study period (* denotes presence)

Species	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
ACRIDIDAE												
<i>Acorypha ferrifer</i> (Walker)	*	*	*									
<i>Acorypha nigrovariegata tibialis</i> (Kirby)	*	*	*	*	*	*	*	*	*	*	*	
<i>Acrida</i> sp.	*	*	*	*	*					*		*
<i>Acrida acuminata</i> (Stål.)	*	*	*					*	*	*	*	*
<i>Acrida bicolor</i> (Thunberg)		*										
<i>Anablepia dregei</i> (Ramme)		*	*						*	*		
<i>Aneuryphymus montanus</i> (Brown)	*							*		*	*	*
<i>Anthermus granosus</i> (Stål.)		*	*	*	*	*		*	*	*	*	
<i>Calliptamulus natalensis</i> (Sjöstedt)	*	*										
<i>Calliptimicus semiroseus</i> (Serville)	*	*	*						*	*	*	*
<i>Cannula gracilis</i> (Burmeister)			*	*	*				*			
<i>Catantops melanostictus</i> (Schaum)	*		*	*				*	*	*	*	*
<i>Coryphosima stenoptera</i> (Schaum)	*	*	*	*							*	*
<i>Crucinotacris cruciata</i> (I. Bolivar)	*	*	*	*	*	*	*	*	*	*	*	*
<i>Cyrtacanthacris aeruginosa</i> (Stoll.)		*	*	*	*	*						
<i>Dirshia abbreviata</i> (Brown)		*	*	*								
<i>Eupropracris cylindricollis</i> (Schaum)											*	
<i>Eyprepocnemis plorans</i> (Charpentier)	*	*	*	*								
<i>Faureia milanjica</i> (Karsch)				*	*	*			*	*		
<i>Faureia rosea</i> (Uvarov)					*	*		*				
<i>Gastrimargus crassicornis</i> (Saussure)	*	*	*	*	*						*	*
<i>Gastrimargus determinatus vitripennis</i> (Walker)	*	*	*	*							*	*
<i>Gastrimargus drakensbergensis</i> (Ritchie)	*	*	*	*								
<i>Gymnobothrus temporalis temporalis</i> (Stål.)			*	*				*		*		*
<i>Heteracris herbacea</i> (Serville)	*	*	*									
<i>Heteroptermis guttifera</i> (Kirby)	*	*	*	*	*							
<i>Macheridia bilineata</i> (Stål.)			*		*	*	*	*	*	*	*	*
<i>Ornithacris cyanae</i> (Stoll.)				*	*				*	*	*	*
<i>Orthochtha rosacea</i> (Walker)	*	*	*	*	*	*		*	*	*	*	*
<i>Orthochtha zuluensis</i> (Popov and Fishpool)	*	*	*						*	*		*
<i>Orthochtha dasycnemis</i> (Gerstaecker)		*	*									
<i>Oxya hyla hyla</i> (Serville)	*	*	*	*								*
<i>Paracinema tricolor</i> (Thunberg)	*											*
<i>Parga xanthoptera</i> (Stål.)	*			*	*		*	*	*		*	
<i>Phaeocatantops sulphurius</i> (Walker)		*	*	*	*	*		*	*	*	*	*
<i>Pnorisa squalus</i> (Stål.)		*	*									
<i>Rhaphotittha cephalica</i> (I. Bolivar)		*	*									
<i>Scintharista rosacea</i> (Kirby)	*	*	*									
<i>Vittacatantops botswana</i> (Jago)	*		*	*	*	*		*	*	*	*	*
LENTULIDAE												
<i>Eremidium basuto</i> (Brown)	*		*		*	*	*	*	*	*	*	*
<i>Lentula minuta</i> (Dirsh)	*	*	*							*		*
<i>Lentula obtusifrons</i> (Stål.)	*	*	*	*					*	*	*	*
<i>Qachasia fastigiata</i> (Dirsh)	*	*	*	*	*					*	*	*
PAMPHAGIDAE												
<i>Transvaaliana draconis</i> (Brown)	*											*

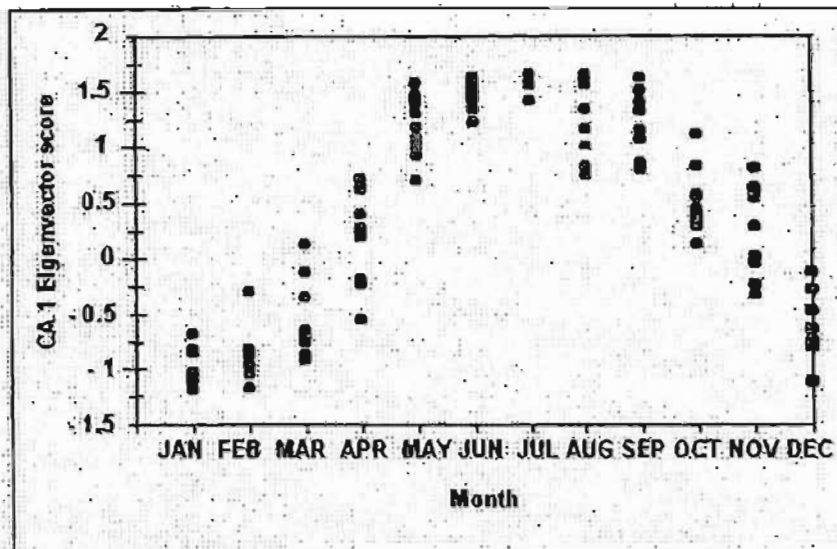
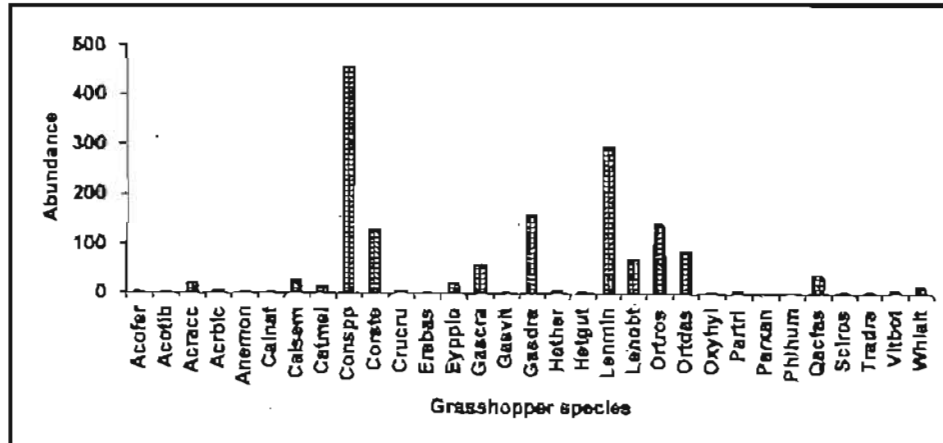
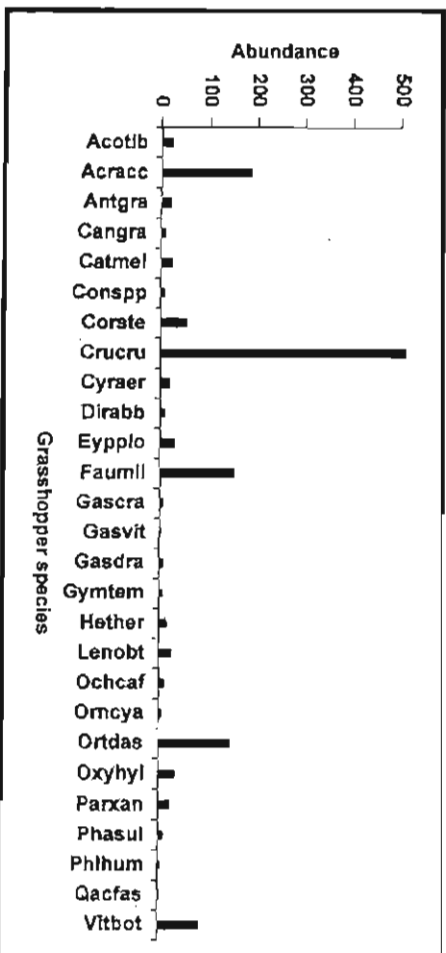
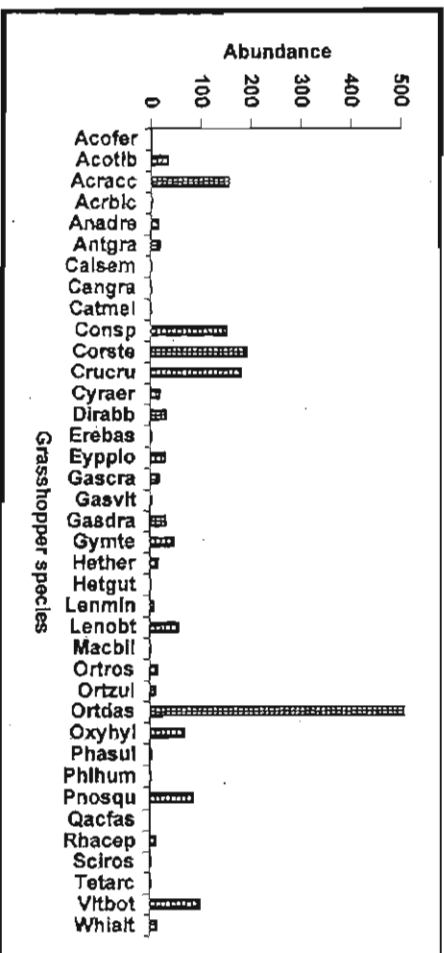
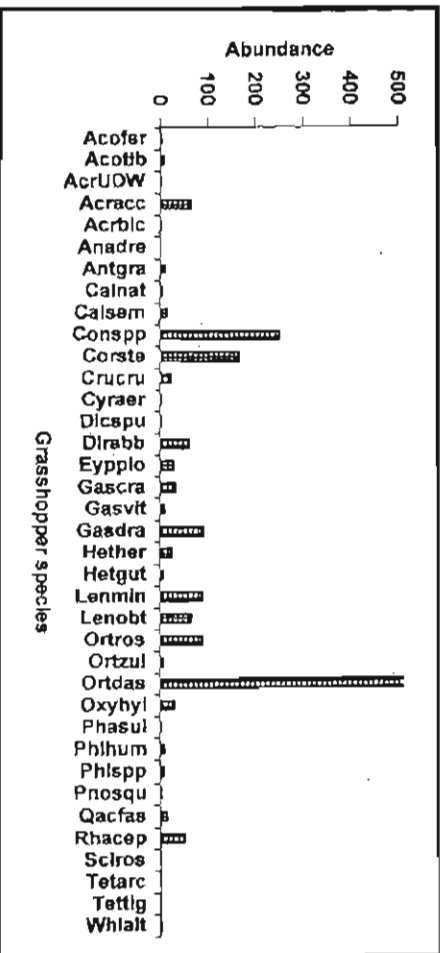


Figure 14. Eigenvector scores on the first axis of a Correspondence Analysis (CA) versus months to show grasshopper species compositional change over one year for all sites (The dots represent each site for the year of sampling).

There was a marked separation of months into two periods, namely, a 'summer' period and a 'winter' period (Figs 13, 14 and 16). Therefore, it was important to separate the data to illustrate how the environmental variables were affecting species composition during the two different periods of the year.

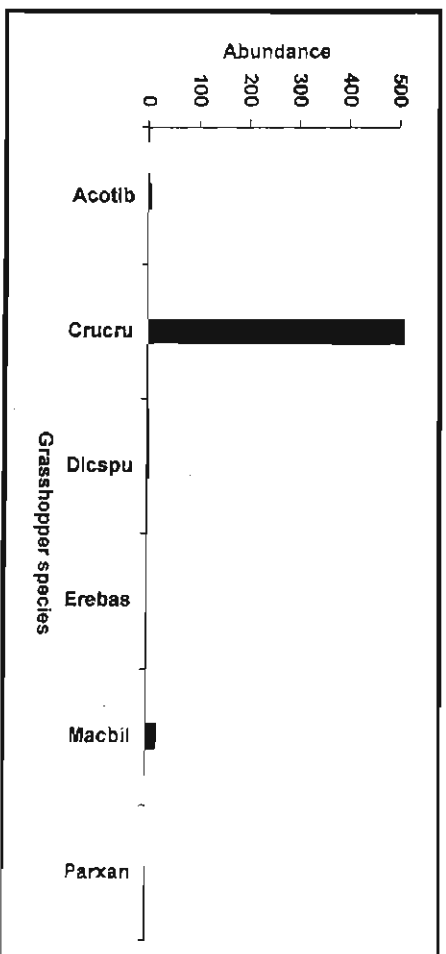
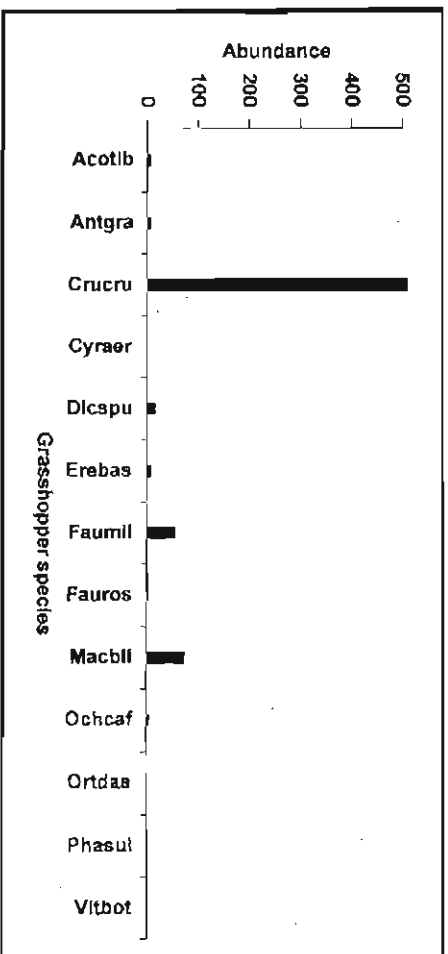
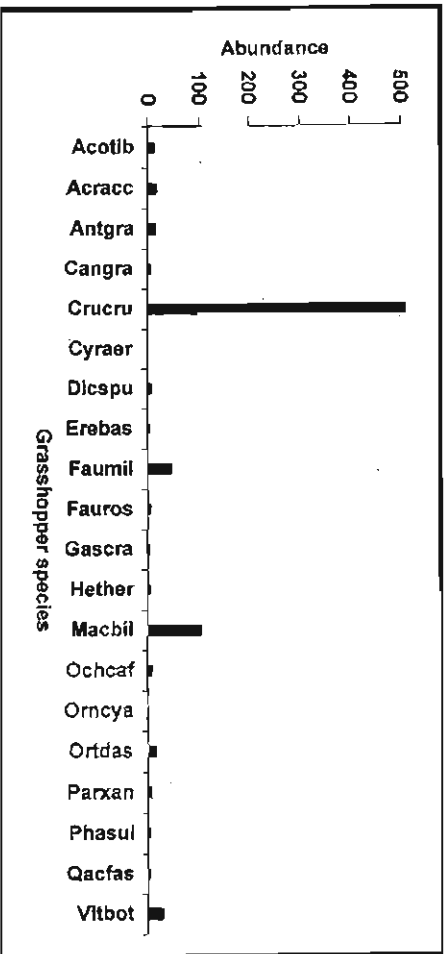


(a) January

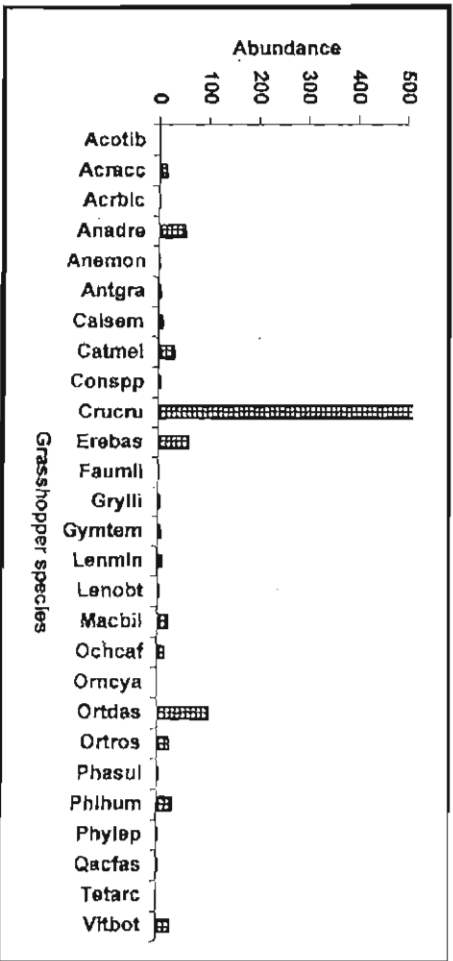


BIOTIC INDICATORS OF GRASSLAND CONDITION IN KWAZULU-NATAL WITH MANAGEMENT RECOMMENDATIONS

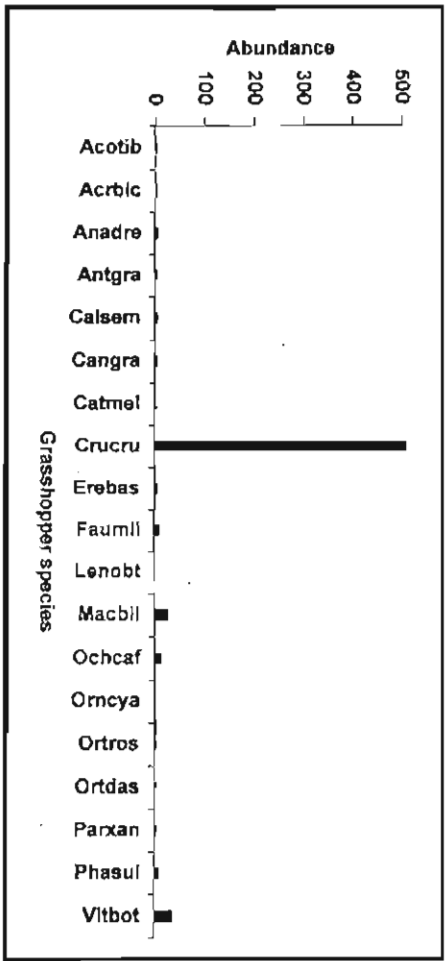
BIOTIC INDICATORS OF GRASSLAND CONDITION IN KWAZULU-NATAL WITH MANAGEMENT RECOMMENDATIONS



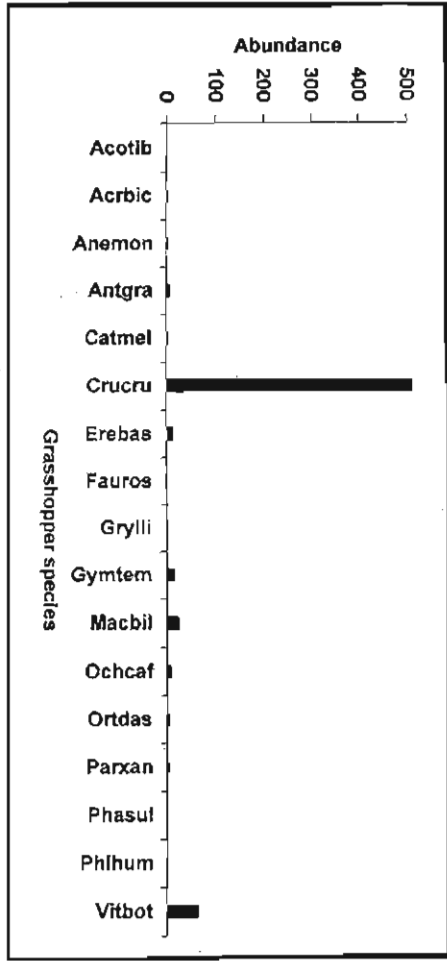
(j) October

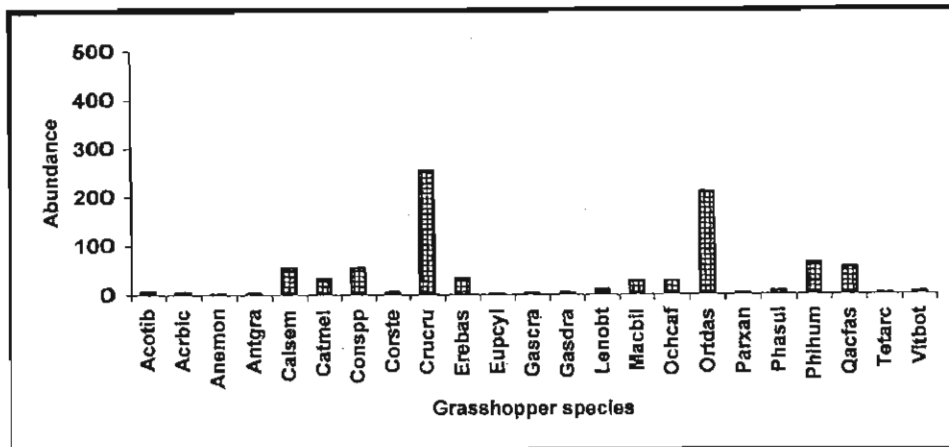


(i) September

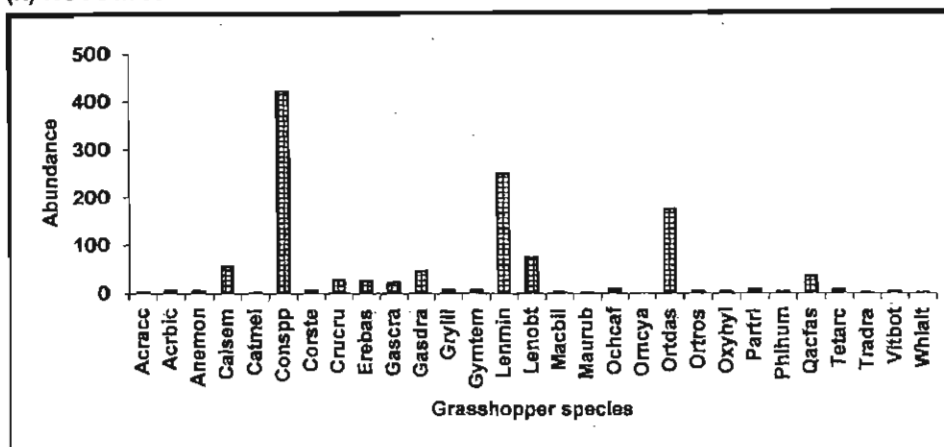


(h) August





(k) November



(l) December

Figure 15 (a) to (l). Grasshopper species recorded across all of the ten sites for each month of the study period. (Summer months are represented by hatched bars and winter by full bars). (Refer to Table 6 for grasshopper species codes).

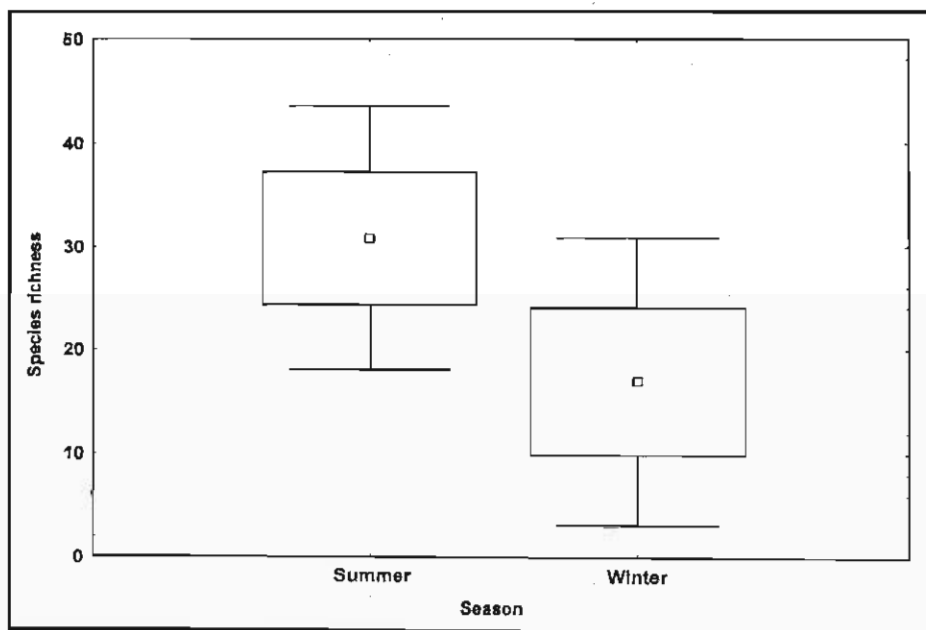


Figure 16. An analysis of variance comparing the summer and winter grasshopper species richness for the twelve-month study period.

Figure 16. An analysis of variance comparing the summer and winter grasshopper species richness for the twelve-month study period.

Table 15. Grasshopper species richness when the two sampling periods were compared summer = October to March, winter = April to September. (***) Denotes significant difference at $p < 0.05$)

Sampling period	df	F-statistic	p-value
summer/winter	1	11.16	0.00***

An analysis of variance that used the divisions of months from the correspondence analysis and the Bray-Curtis Analysis to determine species richness across the two sampling periods was highly significant at $p < 0.05$ ($F = 11.16$ and $p = 0.00$).

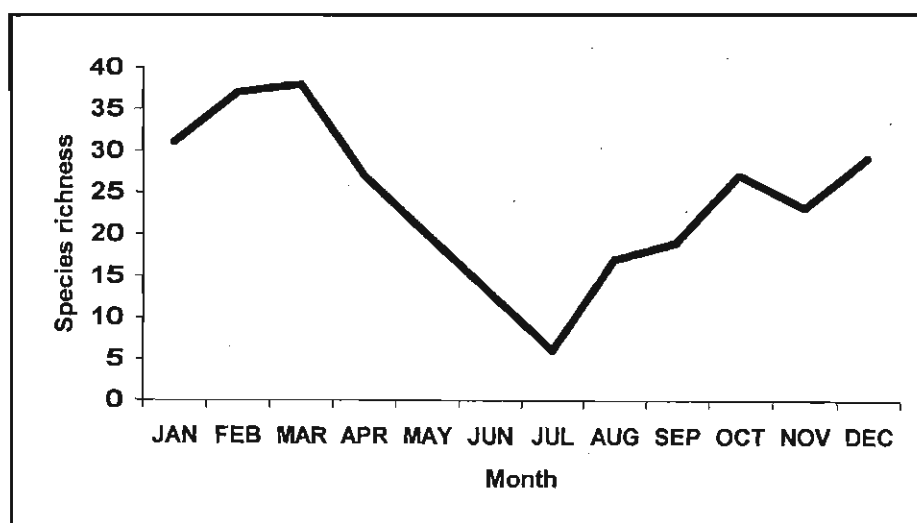


Figure 17. Species richness of grasshoppers during the whole study period.

Table 16. Grasshopper diversity indices, evenness and species richness for each of the 12 months of the study period (data log transformed)

Month	Shannon's	Evenness (Pielou's J)	Abundance	Species richness
JAN	0.975	0.653	1540	31
FEB	1.018	0.649	1856	37
MAR	0.960	0.608	2382	38
APR	0.589	0.411	2970	27
MAY	0.297	0.228	2014	20
JUN	0.241	0.216	1339	13
JUL	0.059	0.075	1035	06
AUG	0.270	0.219	1039	17
SEP	0.325	0.254	760	19
OCT	0.641	0.448	1261	27
NOV	0.961	0.706	865	23
DEC	0.918	0.628	1212	29

Grasshopper evenness was markedly different when months of the year were compared. One species or a suite of species dominated seven months of the study

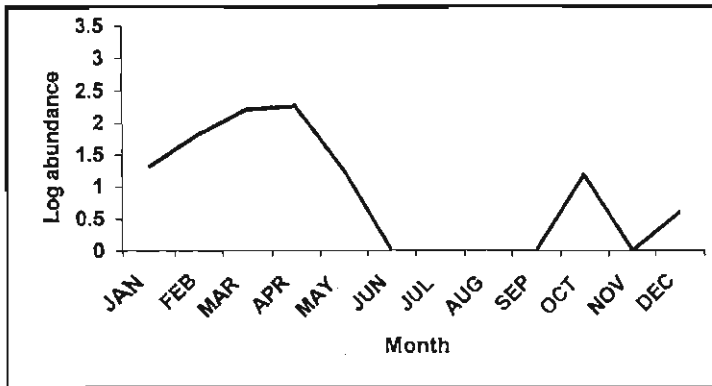
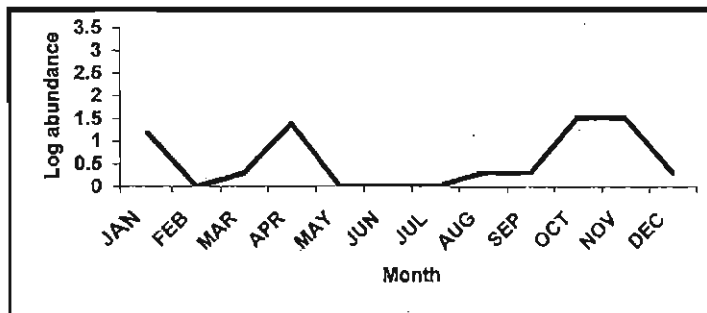
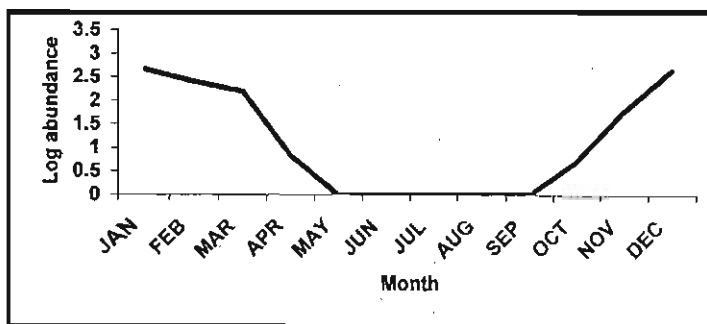
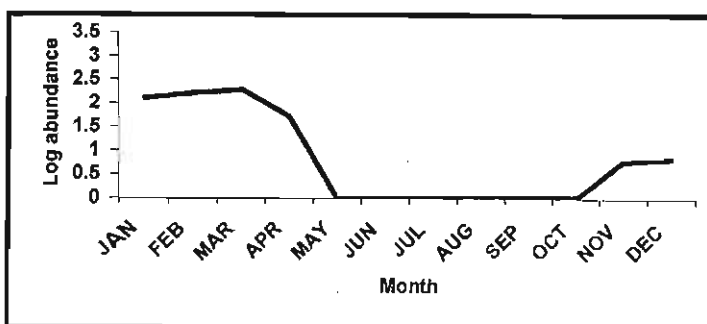
period. The months were predominantly winter months (May through August) and three months were in the Spring/Autumn time of the year (April, September and October) (Table 16 and Figure 17). In the winter months *C. cruciata* was very abundant and there were also fewer species with the onset of colder weather.

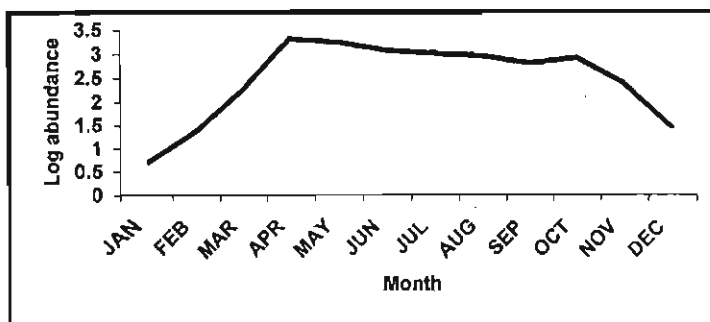
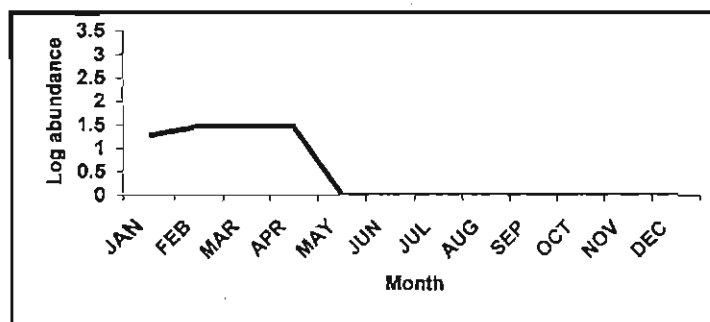
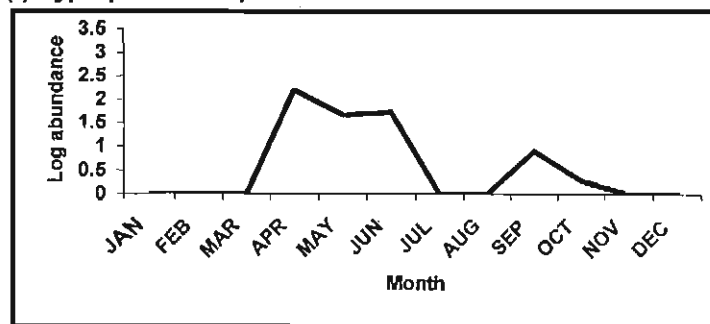
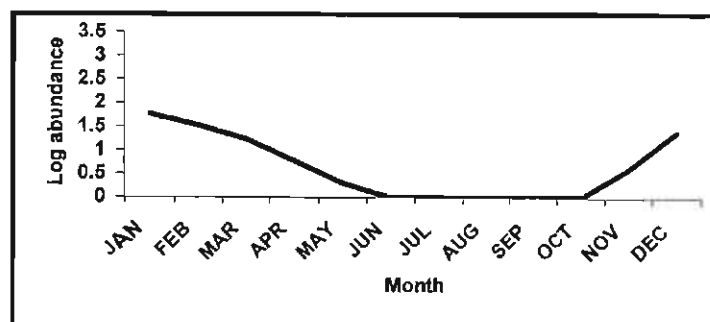
3.7.2. Individual grasshopper responses to seasonality

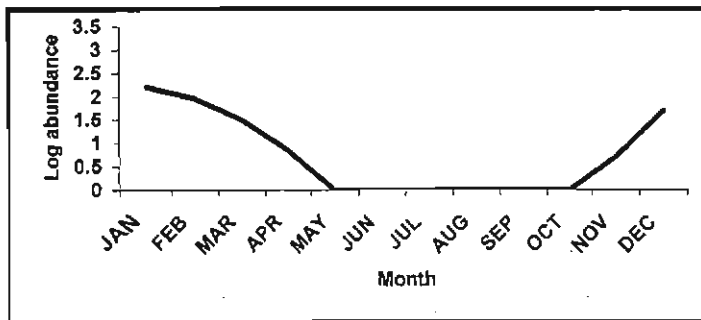
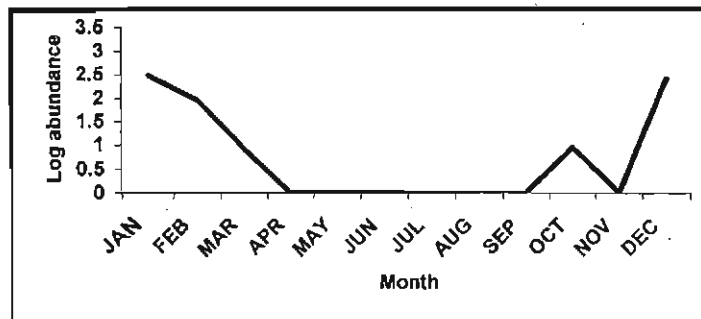
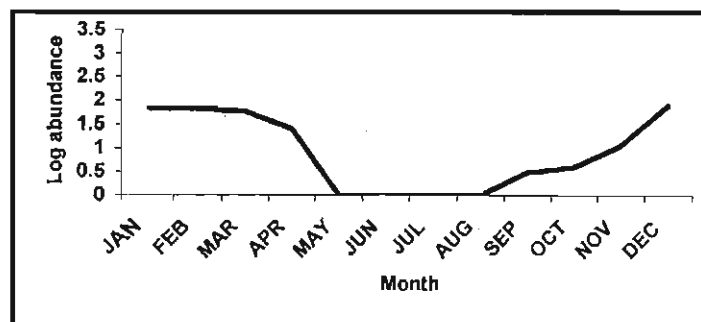
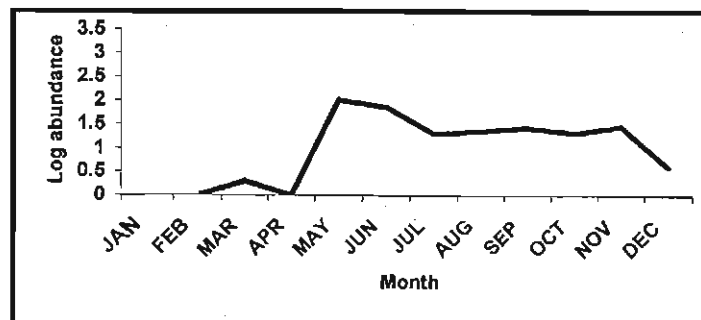
13 species had more than 20% of their variation explained by Axis 1 and 2 and were plotted to show their seasonal cycle. The species abundances were log transformed for the 13 species as some species had very high abundances. Of the 13, one species, *C. melanostictus*, showed two distinct periods when they were present in the system. This therefore implies that this particular species had two hatchings in a year (bivoltine), spring and autumn, with eggs dormant during the extremes of summer and winter (Fig. 18, (b)). The other species that appeared to have two hatchings was *Faureia milanjica*, but its numbers were affected by the time at which the grassland was burnt. The numbers increased in September due to recruitment from surrounding unburnt grasslands.

Nine species showed a clear summer-dominated cycle, having recorded high abundances during this period and being absent during the winter, starting in April or May. The species that showed this pattern are *Acrida acuminata*, *Conocephalus* sp., *Coryphosima stenoptera*, *Eyrepocnemis plorans*, *Gastrimargus crassicollis*, *Gastrimargus drakensbergensis*, *Lentula minuta*, *Lentula obtusifrons*, *Qachasia fastigiata* and *Orthochtha dasycnemis* (Fig. 18, (a), (c), (d), (f), (i), (j), (k), (l), (m) and (n)).

Macheridia bilineata and *Cruciotacris cruciata* were adult in winter, appearing in May-December (*M. bilineata*), or all year (*C. cruciata*), but abundance was highest in the winter period (Fig. 18, (e) and (h)).

(a) *Acrida acuminata*(b) *Catantops melanostictus*(c) *Conocephalus* sp.(d) *Coryphosima stenoptera*

(e) *Crucinotacris cruciata*(f) *Eyprepocnemis plorans*(g) *Faureia milanjica*(h) *Gastrimargus crassicornis*

(i) *Gastrimargus drakensbergensis*(j) *Lentula minuta*(k) *Lentula obtusifrons*(l) *Macheridia bilineata*

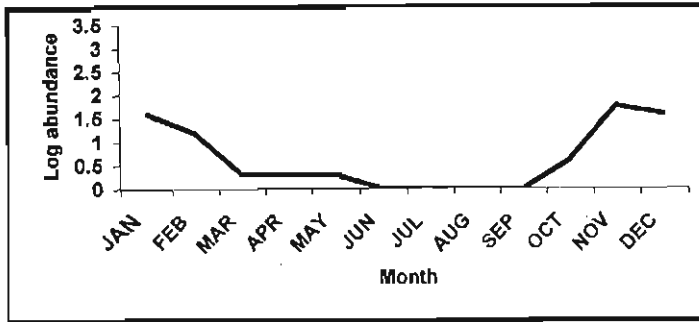
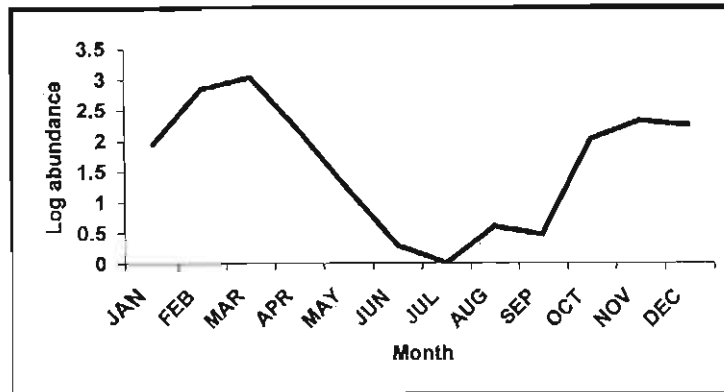
(m) *Qachasia fastigiata*(n) *Orthochtha dasyncnemis*

Figure 18 (a) to (n). Seasonal variation in species that have a cumulative fit of greater than 20% of their variation explained for Axis 1 and 2. All data have been log-transformed.

3.7.3. Butterflies

A month-by-month Bray Curtis Cluster Analysis (Fig. 19) showed how, two large clusters were represented (May to September) and (October to April). This distribution of months into these two large clusters mirrors the situation for grasshoppers, with a distinct 'summer' and 'winter' period.

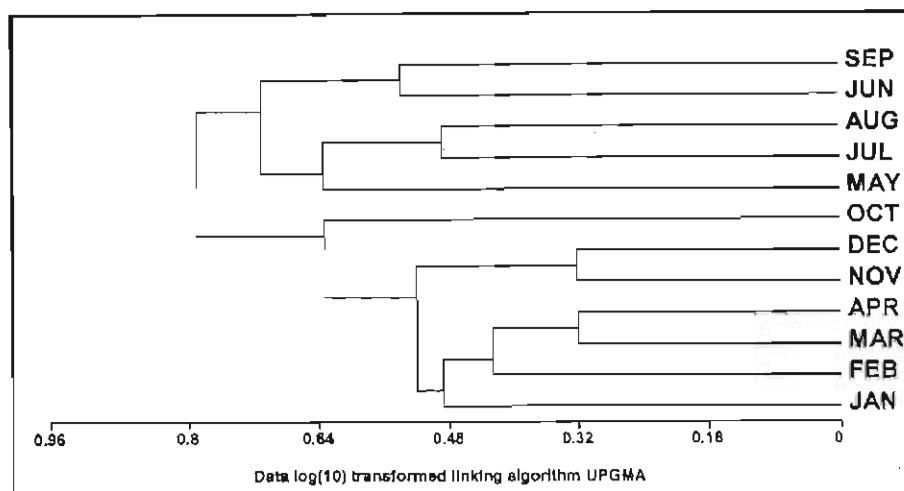


Figure 19. Butterfly similarity across months using a Bray-Curtis Cluster Analysis. (Data were combined and log transformed).

All months had an even distribution. The most even distribution was shown by May and July, but species richness was only six and three, with the most uneven months, March and November with 0.768 and 0.736 and high species richness 17 and 12 respectively (Table 17).

Correspondence Analysis (CA) (Fig. 20) showed no definite groupings of months into sampling seasons or periods (only the centroid values were plotted for each month). 11 of the 12 months clustered together around the midpoint of Axis 1 and 2. The 12th month was June an outlier with a centroid value of 0.12 and 3.31 for Axis 1 and 2 respectively. The Eigen Values for Axis 1 and 2 were 0.35 and 0.23 respectively, with the sum of all the Eigen Values 1.35. The cumulative percent variance explained by Axis 1 and 2 was 43.2%.

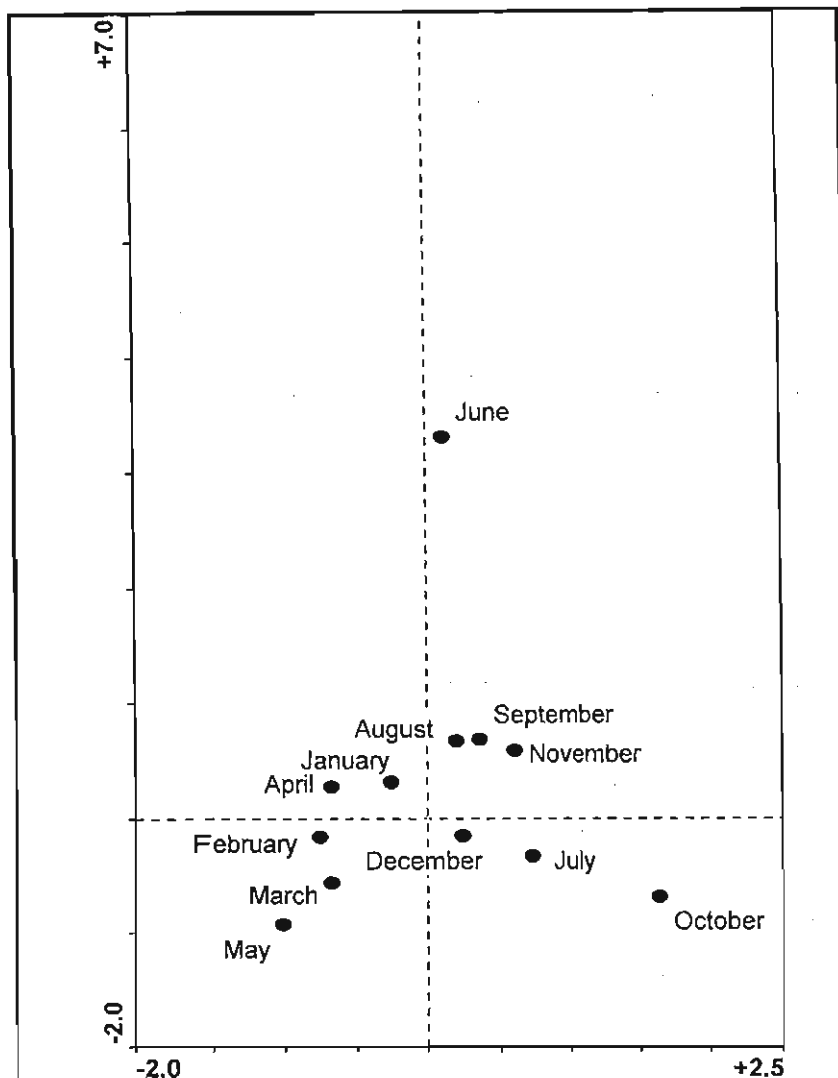


Figure 20. Correspondence Analysis (CA) of month distribution according to the butterfly species for the 10 sites.

The eigenvector scores from the Correspondence Analysis (CA) were plotted against month (Fig. 21). Each dot represents a site for each of the 12 months. The sites are fairly uniformly distributed across the months except for October that showed large dispersal along the vertical axis. This signified that species composition varied dramatically for the month of October (10).

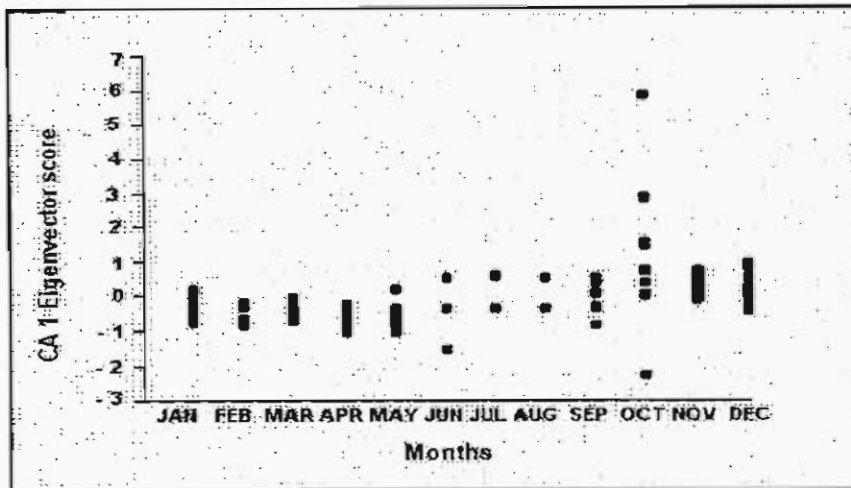
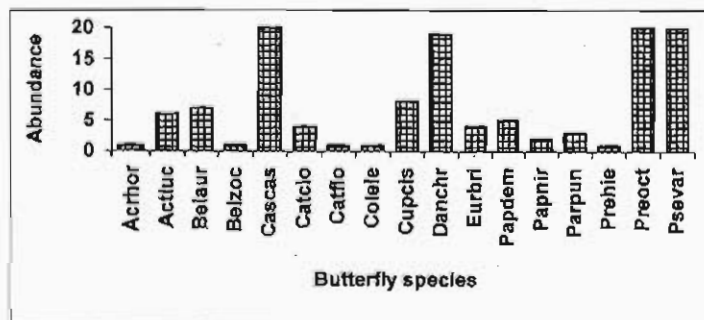


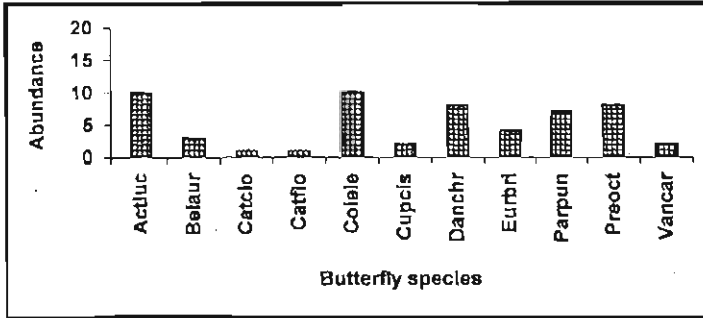
Figure 21. Eigenvector scores versus month for all the sites for butterflies (each dot represents a site).

Table 17. Butterfly diversity indices, evenness and species richness for each of the 12 months of the study period (data were log transformed)

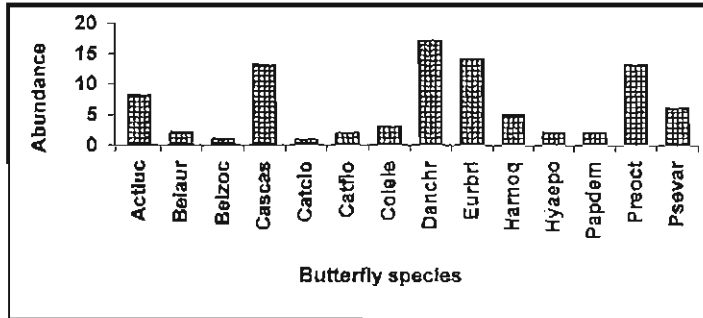
Month	Shannon's	Evenness (Pielou's J)	Abundance	Species richness
JAN	0.791	0.936	81	07
FEB	0.937	0.900	56	11
MAR	0.945	0.768	165	17
APR	0.993	0.866	89	14
MAY	0.736	0.946	10	06
JUN	0.649	0.928	8	05
JUL	0.452	0.946	4	03
AUG	0.276	0.918	3	02
SEP	0.781	0.865	21	08
OCT	0.967	0.896	69	12
NOV	0.794	0.736	101	12
DEC	1.155	0.903	91	19



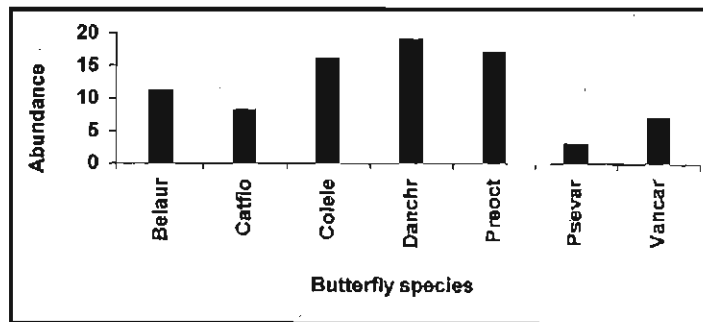
(a) January



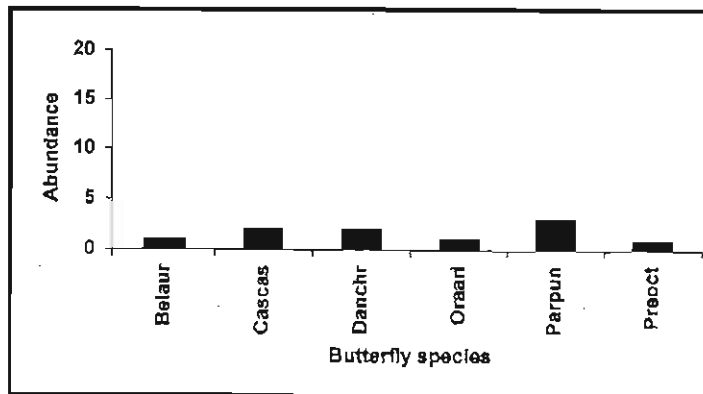
(b) February



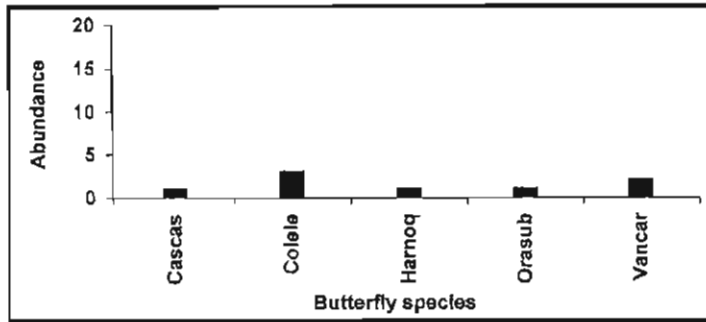
(c) March



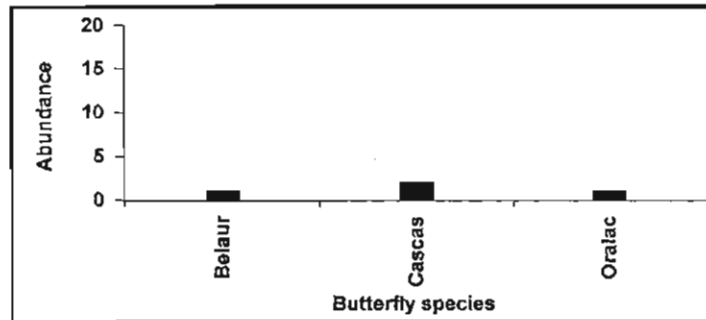
(d) April



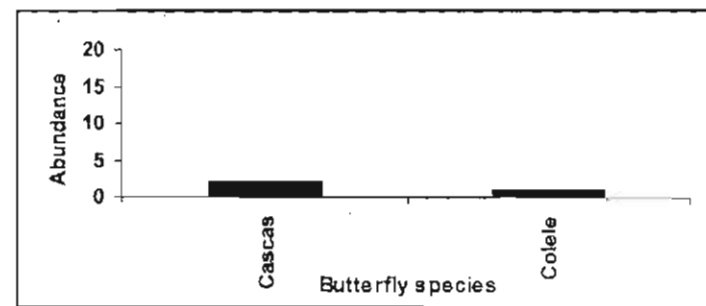
(e) May



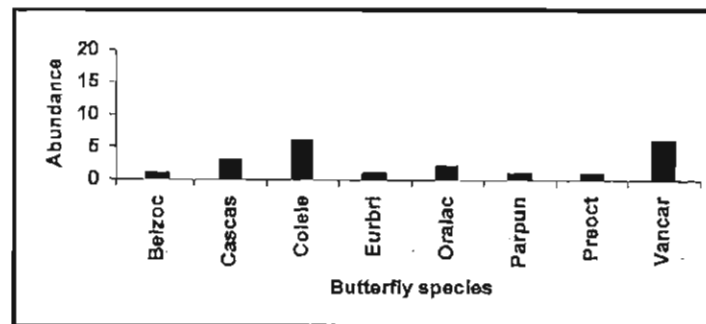
(f) June



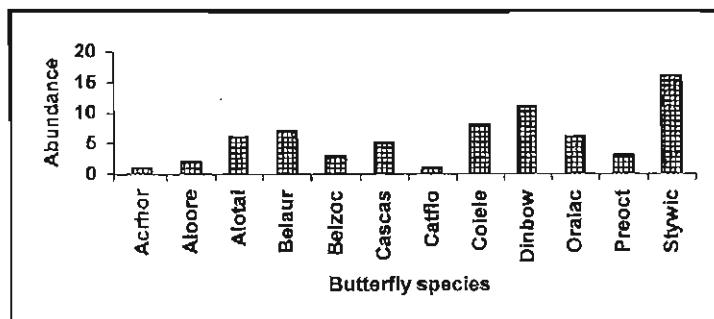
(g) July



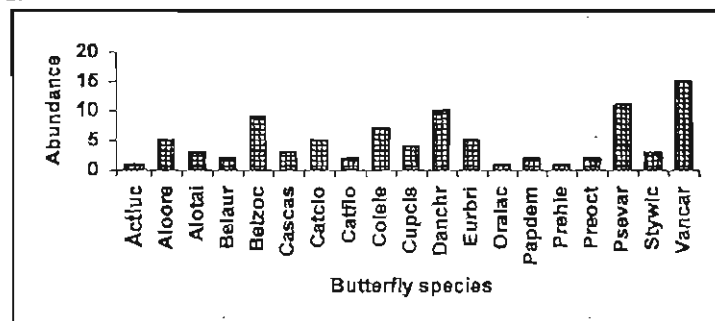
(h) August



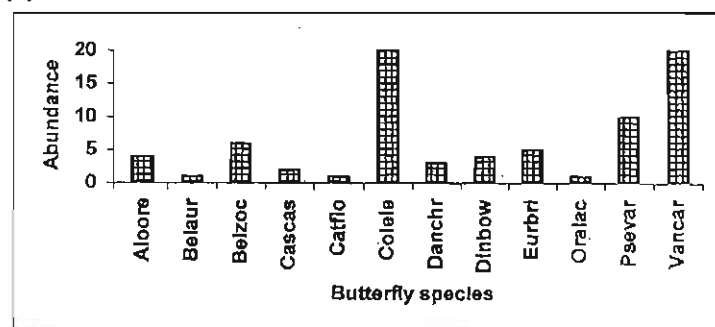
(i) September



(j) October



(k) November



(l) December

Figure 22 (a) to (l). Butterfly abundances for species in each month of sampling. (Summer months indicated by hatched bars and winter by full bars). (Refer to Table 7 for butterfly species codes).

Table 18. A presence absence table of butterfly species for the 12-month study period (* denotes presence)

Species	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
LYCAENIDAE												
<i>Actizera lucida</i> (Trimen)		*	*	*								*
<i>Aloeides oreas</i> (Tite & Dickson)										*	*	*
<i>Aloeides taikosama</i> (Wallengren)										*		*
<i>Cupidopsis cissus</i> (Godart)		*	*									*
<i>Harpedyreus noquasa</i> (Trimen & Bowker)				*		*						
<i>Orachrysops ariadne</i> (Butler)					*							
<i>Orachrysops lacrimosa</i> (Bethune-Baker)							*		*	*	*	*
<i>Orachrysops subravus</i> (Henning)						*						
NYMPHALIDAE												
<i>Acraea horta</i> (Linnaeus)			*							*		
<i>Cassionympha cassius</i> (Godart)			*	*	*	*	*	*	*	*	*	*
<i>Catacroptera cloanthe cloanthe</i> (Stoll.)		*	*	*								*
<i>Danaus chrysippus</i> (Schreber)	*	*	*	*	*						*	*
<i>Dingana bowkeri bowkeri</i> (Trimen)										*	*	
<i>Hyalites eponina eponina</i> (Cramer)				*								
<i>Pardopsis punctatissima</i> (De Boisduval)		*	*		*				*			
<i>Precis hierta hierta</i> (Fabricius)			*									*
<i>Precis octavia</i> Cramer)	*	*	*	*	*				*	*		*
<i>Pseudonympha varii</i> (Van Son)	*		*	*							*	*
<i>Stygionympha wichgrafi wichgrafi</i> (Van Son)										*		*
<i>Vanessa cardui</i> (Linnaeus)	*	*				*			*		*	*
PAPILIONIDAE												
<i>Papilio demodocus</i> (Esper)			*	*								*
<i>Papilio nireus lyaeus</i> (Linnaeus)			*									
PIERIDAE												
<i>Belenois aurota</i> (Fabricius)	*	*	*	*	*		*			*	*	*
<i>Belenois zochalia zochalia</i> (De Boisduval)			*	*					*	*	*	*
<i>Catopsilia florella</i> (Fabricius)	*	*	*	*						*	*	*
<i>Colias electo</i> (Fabricius)	*	*	*	*		*		*	*	*	*	*
<i>Eurema brigitta brigitta</i> (Stoll.)		*	*	*					*		*	*
Species richness	7	11	17	14	6	5	3	2	8	12	12	19

Fig. 22 (a) to (l) shows species richness and species abundance over the whole study period. The most species-rich month was December with 70% of the species, and the most species-poor month (August) with 7% of the species (Table 18). Monthly distributions related to species composition for all ten sites were similar with December, January, February and March clustering together, and the remaining eight months showed an association with one another. This pattern was the general representation of what was occurring across the ten sites, with slight variations occurring at the individual sites (Fig 24).

3.8. Correspondence analysis

3.8.1. Grasshopper species composition

A Correspondence Analysis (CA) (Fig. 23) showed the variation amongst species for all sites across all months. The data were combined to reduce redundancy. The axes represent the summed abundance of all species across all sites and months. The species that were closely clustered together are species which were recorded in similar abundances. There are two apparent clusters and a number of outliers, with the very common species, namely, *C. cruciata*, *G. crassicornis*, *G. Drakensbergenis*, *O. dasycnemis* and *Concephalus* sp. clustering together. The other cluster comprised of less common species that were recorded in relatively high abundance, namely, *L. minuta*, *L. obtusifrons*, *Vittacatantops botswana* and *Eyprepocnemis plorans*. The outliers were present as they were singletons or rarities, and were *D. spumans*, *P. leprosus*, Tettigoniidae sp., *T. draconis*, *E. cylindricollis*, *Acrida* UDW and *M. rubromata*. Eigen values for Axis 1 and 2 were 0.177 and 0.142 respectively. The sum of all Eigen values was 0.818. The cumulative percentage variance explained by Axis 1 and 2 was 39%.

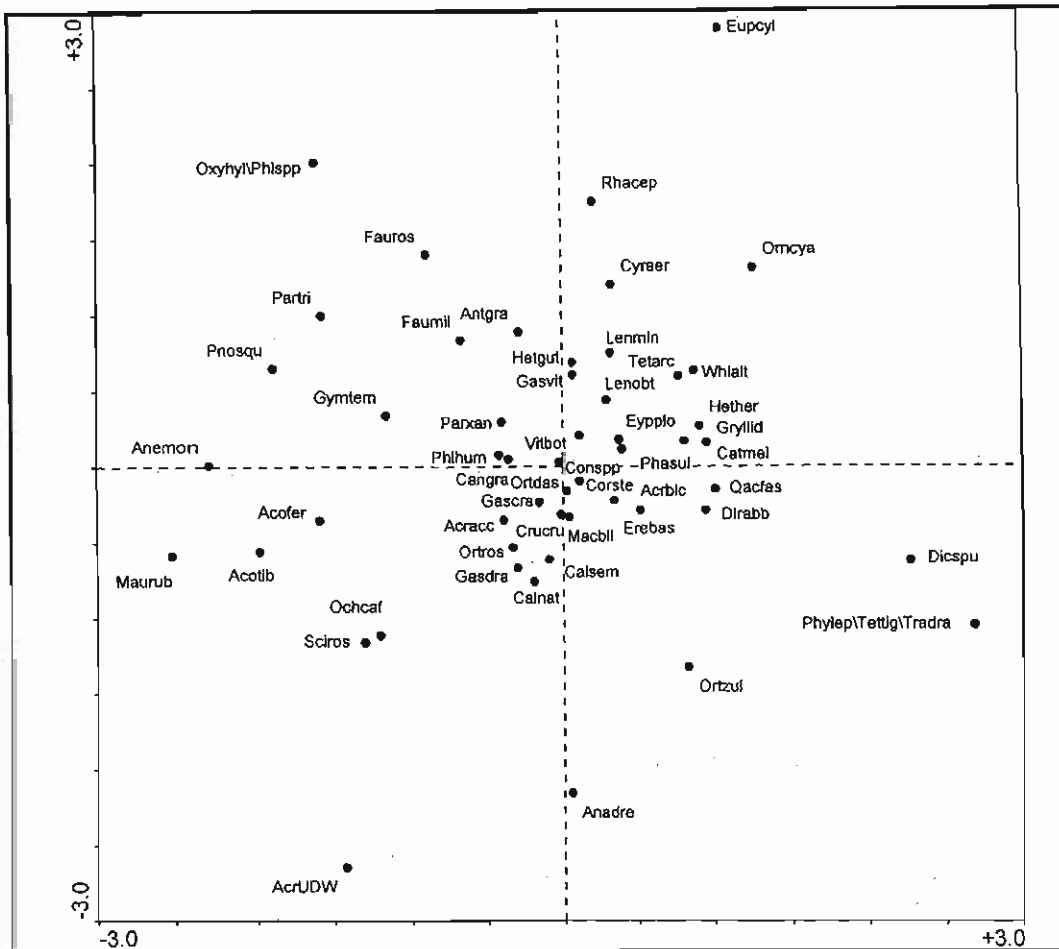
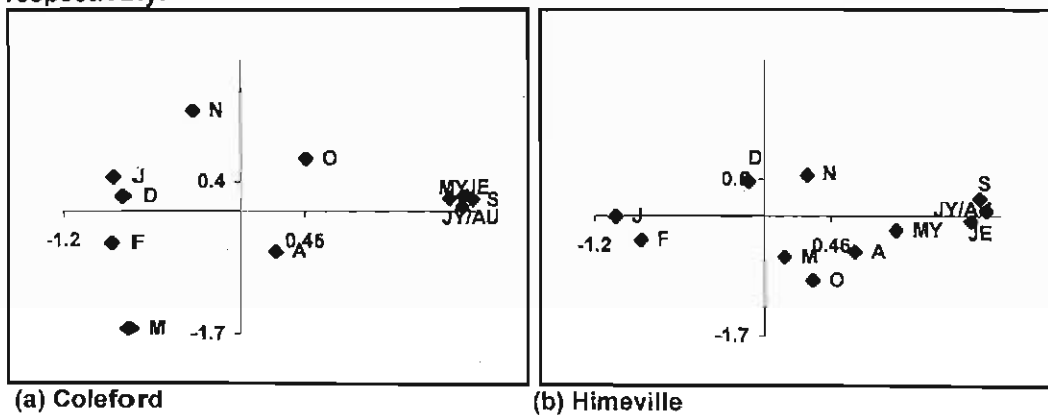


Figure 23. Correspondence Analysis (CA) for grasshopper species composition for all sites across all months. (Data were combined and log transformed). Refer to Table 6 for grasshopper species codes. The Eigen values for Axis 1 and 2 were 0.177 and 0.142 respectively.



(a) Coleford

(b) Himeville

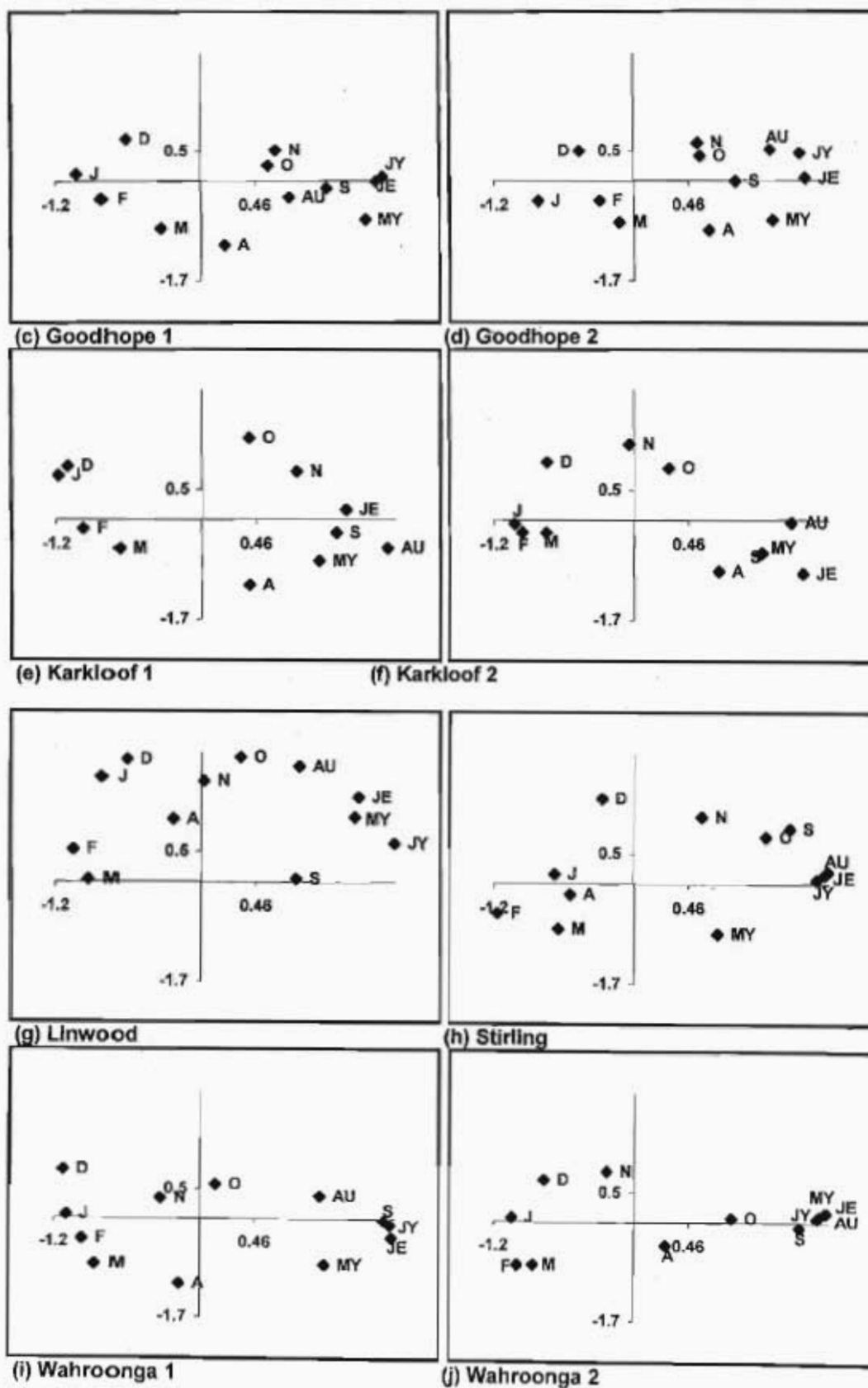


Figure 24. The Correspondence Analysis (CA) for the ten sites across the 12-month study period. (Only centroid values are shown for each month). (J= January, F= February, M=

March, A= April, MY= May, JE= June, JY= July, AU= August, S= September, O= October, N= November, D= December).

3.8.2. Butterfly species composition

Correspondence Analysis (CA) (Fig. 25) shows the species distribution of butterflies for the whole study period. The Eigen values were 0.227 and 0.168 for Axis 1 and 2 respectively. The cumulative percent variance explained by Axis 1 and 2 was 44.7%.

There were two species clusters. One cluster comprised of six species, with the other cluster being made up of 18 species. Three species plotted were outliers, namely, *O. ariadne*, *P. varii* and *D. bowkeri bowkeri*.

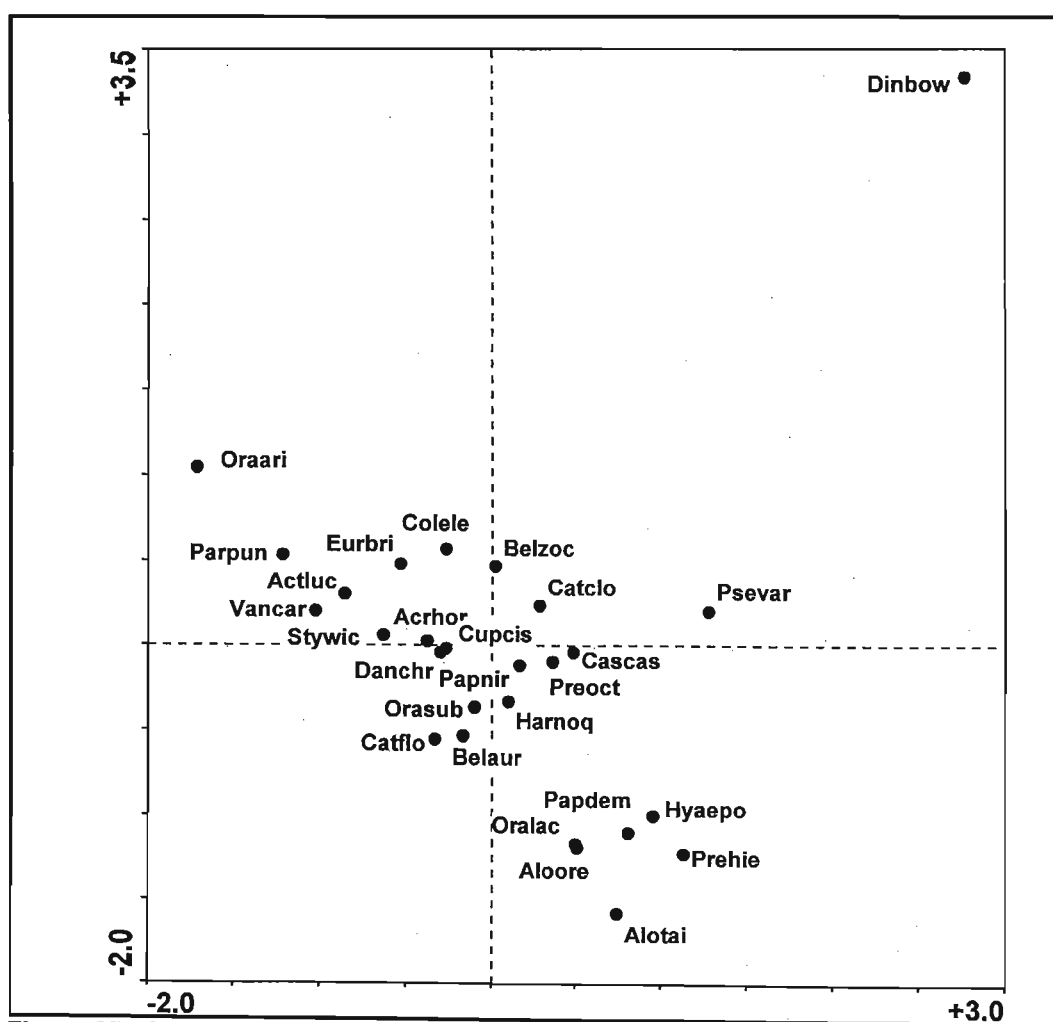


Figure 25. Correspondence Analysis (CA) of butterflies for the whole study period. Refer to Table 7 for butterfly species codes. The Eigen values were 0.227 and 0.168 for Axis 1 and 2 respectively.

3.8.3. Environmental variables

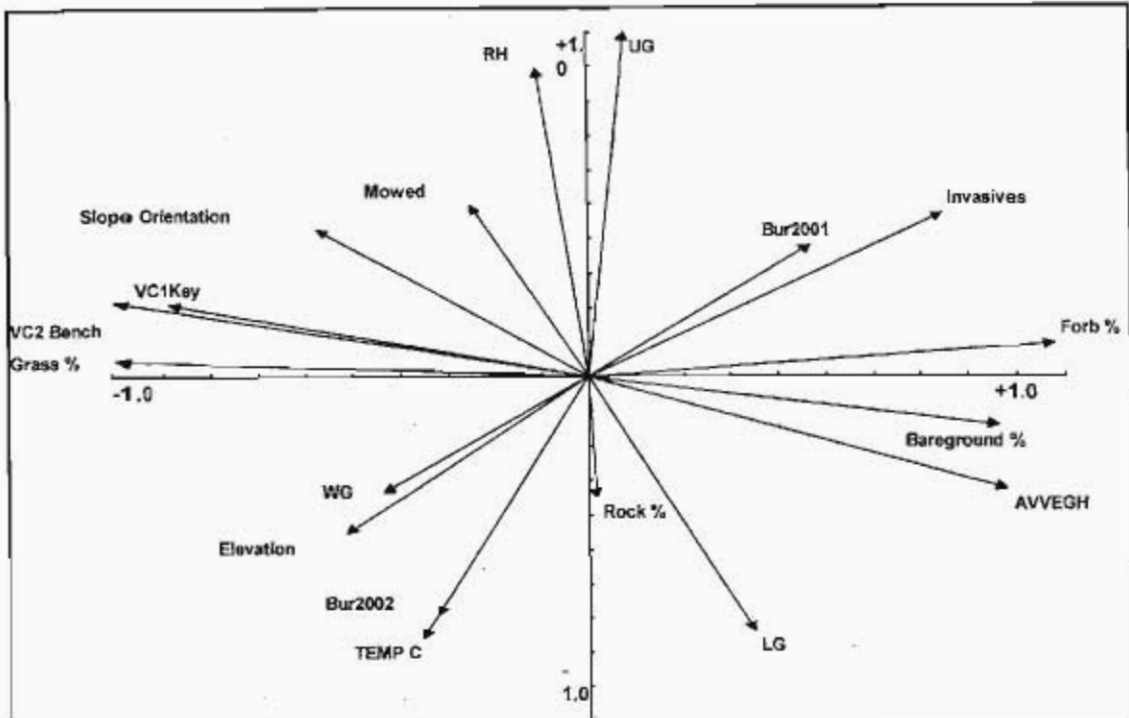


Figure 26. Principal Component Analysis (PCA) of the environmental variables measured illustrating colinearity between the environmental variables. (Abbreviations as per Table 19).

A Principal Component Analysis (PCA) was undertaken to show the colinearity of the environmental variables that were measured (Fig. 26). Many of the variables were omitted as their effects on the species were mirrored in other variables (i.e. colinearity).

Table 19. The codes and explanations of the codes used in the Canonical Correspondence Analysis, and variables omitted as their affects are mirrored by other variables (* indicates omitted)

Codes used	Explanation of Codes	Omitted
Temp C	Average Temperature °C	
RH	Average Relative Humidity	*
Slope	The orientation of the slope (N,S,W,E)	
Elevation	Elevation (m)	
VC1 Key	Veld condition score using Key grass species (Camp, 1998)	*
VC2 Bench	Veld condition score Benchmark grass species (Camp, 1998)	*
AVVEGH	Average vegetation height	*
WG	Wildlife Grazed	
LG	Livestock Grazed	
UG	Not utilized	*
Bur 2001	Burnt in 2001	
Bur 2002	Burnt in 2002	
Rock %	The proportion of rock at each site	*
Forb %	The proportion of forbaceous plants at each site	*
Bareground %	The proportion of ground with no basal cover	*
Invasives	Invasive plants, alien or indigenous	*
Mowed	The site was subjected to a mowing regime	*
Grass %	The proportion of the grassland comprising grass species	

3.9. Species responses to environmental variables

3.9.1. Grasshoppers

A Canonical Correspondence Analysis (CCA) (Fig. 27) of species versus environmental variables was undertaken to test whether the measured variables were having an effect on the species. The sum of all the Eigen values was 0.382. Axis 1 and 2 explained 69.2% of the variance of the species-environment relationship. Axis 1 was significant only at the 10% level ($F= 1.211$ and $p= 0.060$). The combined Axes were significant at the 5% level ($F= 1.301$ and $p= 0.045$).

There were two basic axes described by the environmental variables. The first important Axis was related to Slope orientation where three species were associated positively with warmer slopes (*D. spumans*, *D. abbreviata* and *Q. fastigiata*) and two species were associated with cooler slopes (*F. rosea* and *A. nigrovariegata tibialis*). The second Axis was most strongly related to Grass (grass cover). Two species were associated with better grass condition, namely, *M. bilineata* and *G. drakensbergensis*. Six species were negatively associated with grass condition preferring lower elevations

and a poorer grass sward (*E. cylindricollis*, *O. cyanae*, *C. aeruginosa*, *R. cephalica*, *L. minuta* and *Conocephalus* sp.). Little compositional variation was due to minimum temperature (Min T).

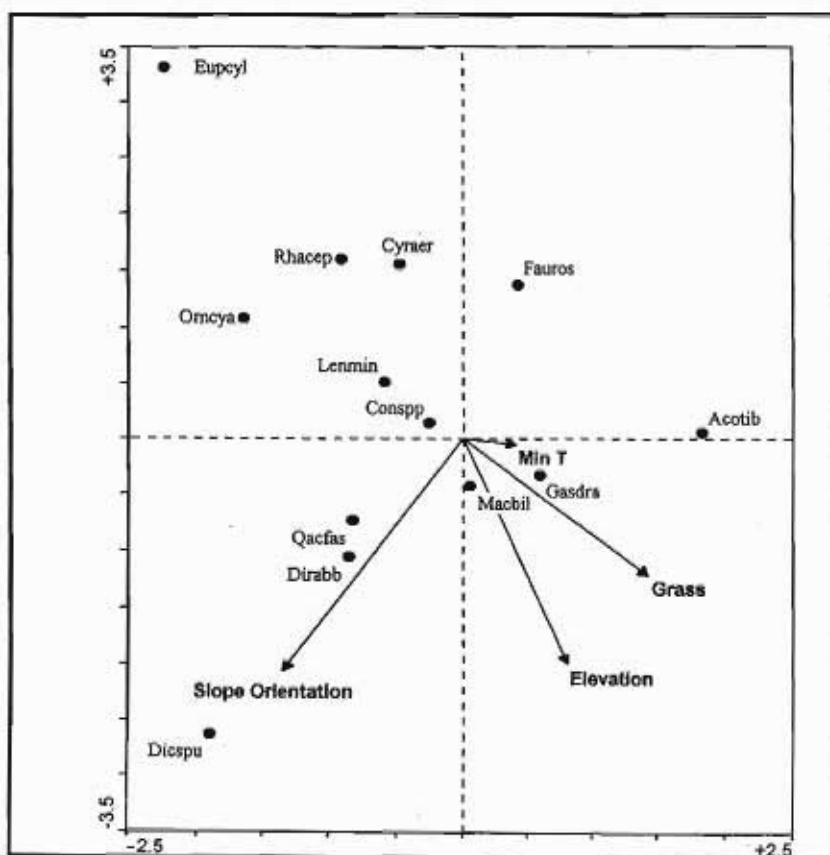


Figure 27. Canonical Correspondence Analysis (CCA) for environmental variables against the sum of the species (Species with 50% variance explained plotted). The combined axes were significant ($F= 1.301$ and $p= 0.045$) (Abbreviations as per Table 19). Refer to Table 6 for grasshopper species codes.

Table 20. The cumulative fit of the species, for the whole study period, versus the environmental variables, as a fraction of the variance, for grasshopper species with greater than 50% variance. Refer to Table 6 for grasshopper species codes

Species code	Axis 1	Axis 2
ACOTIB	0.61	0.62
CONSPP	0.51	0.66
CYRAER	0.06	0.66
DICSPU	0.24	0.65
DIRABB	0.22	0.53
EUPCYL	0.19	0.57
FAUROS	0.06	0.56
GASDRA	0.43	0.54
LENMIN	0.43	0.72
MACBIL	0.03	0.79
ORNCYA	0.36	0.50
QACFAS	0.35	0.59
RHACEP	0.19	0.74

3.9.2. Butterflies

A Canonical Correspondence Analysis (CCA) was undertaken to determine the effect of the measured environmental variables on the sum of the species encountered. Axis 1 and 2 explained 66.9% of the variance of the species-environment relationship. The sum of all the Eigen values was 0.451. Axis 1 was not significant at the 5% level ($F=1.289$ and $p=0.170$). All the axes combined were not significant ($F=1.309$ and $p=0.105$) (Fig. 28).

Two species were positively associated with Min T ($^{\circ}\text{C}$) preferring an environment with a cooler mean temperature; species were *C. florella* and *D. chrysippus*. Two species, namely, *P. varii* and *D. bowkeri bowkeri* were negatively associated with Min T ($^{\circ}\text{C}$), preferring warmer north-facing slopes that received high levels of insulation.

Two environmental variables 'Elevation' and 'Grass' were strongly associated to one another. Three species were negatively associated with this gradient (*A. lucida*, *P. punctatissima* and *O. ariadne*), these species preferring lower elevations and poor veld condition (more forbs and woody vegetation than grass).

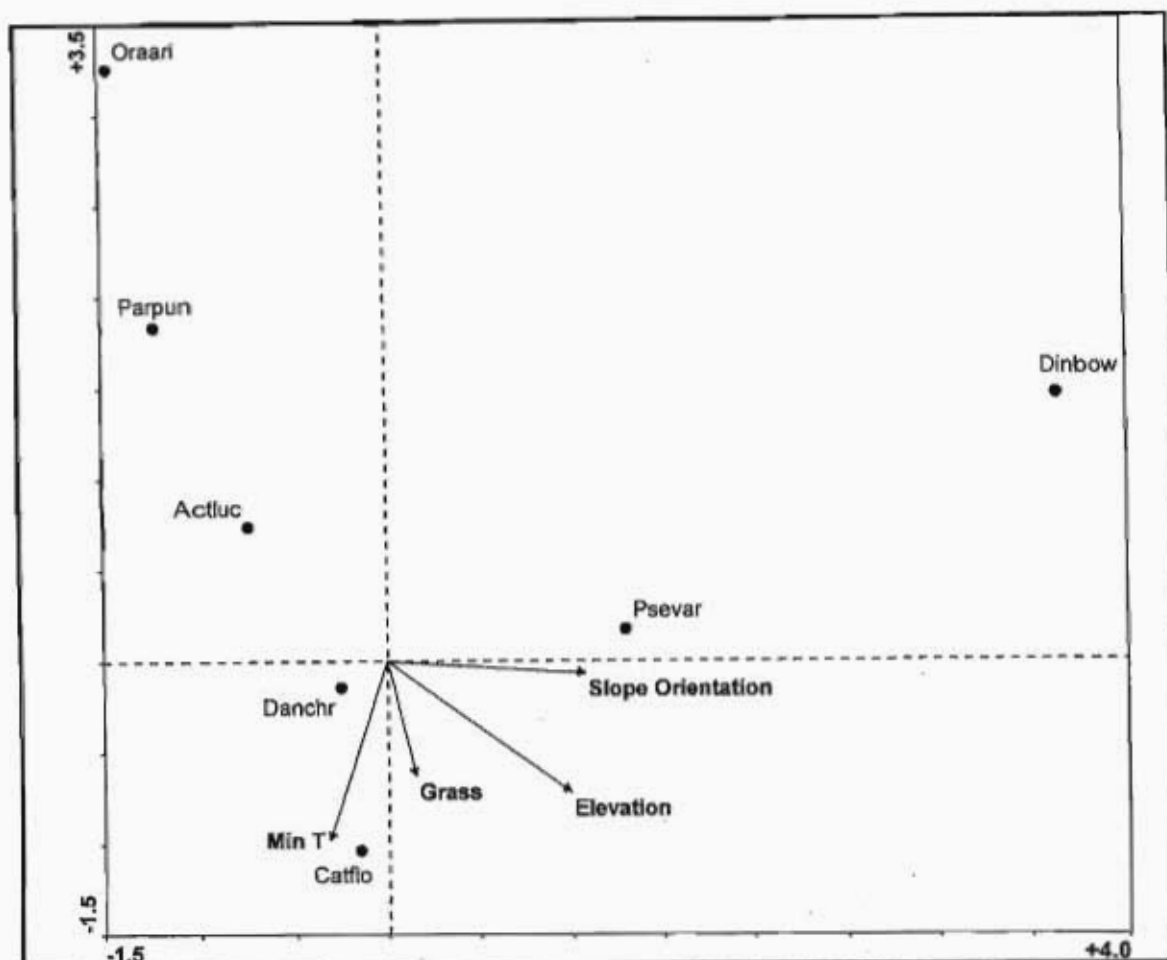


Figure 28. Canonical Correspondence Analysis (CCA) of butterflies versus environmental variables. The combined axes were not significant ($F= 1.309$ and $p= 0.105$) (Only species with $>50\%$ variance explained were plotted). (Grass= overall grass %; Min T= Minimum temperature recorded at each site during the whole study period). (Refer to Table 7 for butterfly species codes).

3.10. Species responses to nominal variables

3.10.1. Grasshoppers

Canonical Correspondence Analysis (CCA) (Fig. 29) of species versus nominal variables was run to show the effect of the measured variables on the species. The sum of all the Eigen Values was 0.431. Axis 1 and 2 explained 55.4% of the variance of the species-environment relationship. Axis 1 was not significant at the 5% level ($F= 0.809$ and $p= 0.600$). The combined Axes test was not significant ($F= 0.809$ and $p= 0.290$). Three species showed a positive association with the nominal variables WG (wildlife grazed) and Highland Sourveld (*A. dregei*, *A. acuminata* and *C. cruciata*). *G. temporalis temporalis*, *P. squalus*, *Phloeonotus* sp. and *O. hyla hyla* were all

associated with LG (livestock grazing). Midlands Mistbelt grassland had four species associated with it, namely, *F. milanjica*, *F. rosea*, *R. cephalica* and *O. cyanae*.

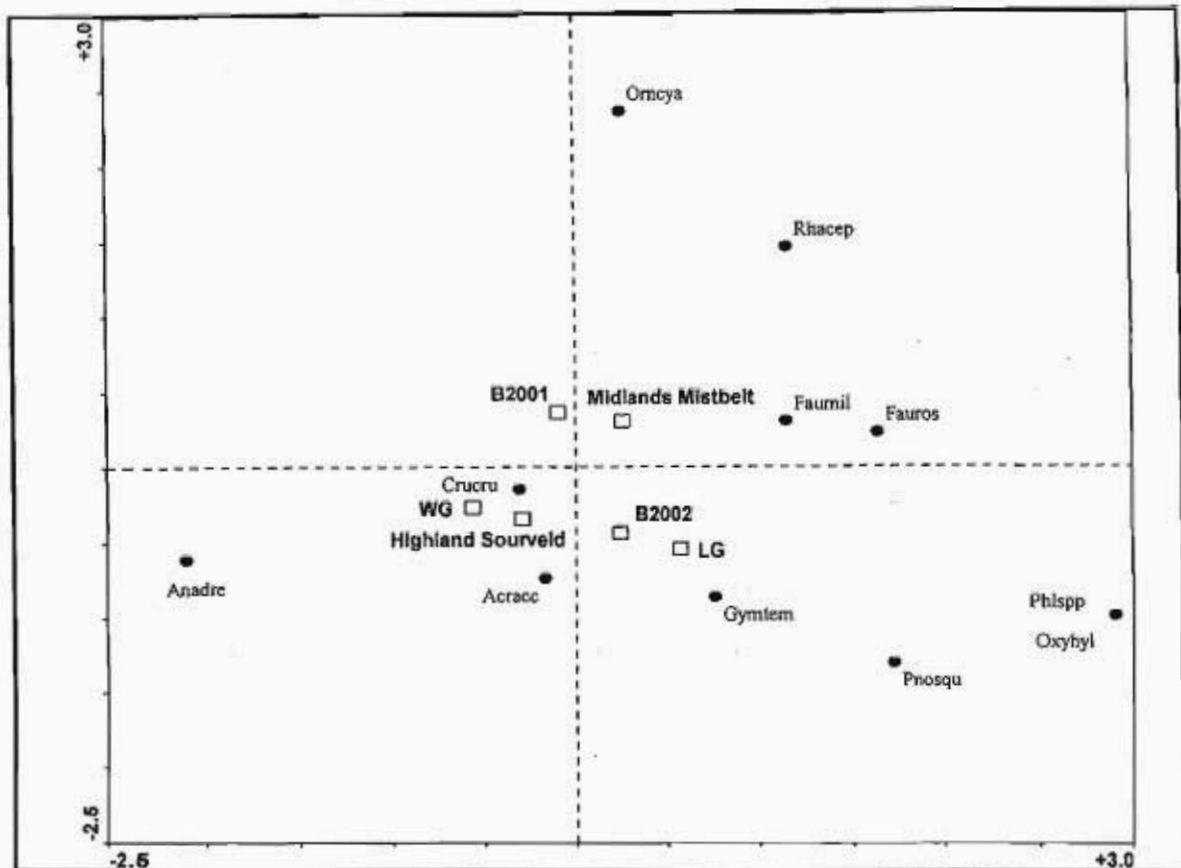


Figure 29. Canonical Correspondence Analysis (CCA) for nominal variables against species (Species with 50% variance explained plotted). Combined axes were not significant ($F=0.809$ and $p=0.290$). (Abbreviations as per Table 19). (Refer to Table 6 for grasshopper species codes).

3.10.2. Butterflies

Canonical Correspondence Analysis (CCA) (Fig. 30) of nominal variables versus species the sum of all the Eigen Values as 0.534, with a cumulative percent variance explained of 52.1% for Axis 1 and 2. Axis 1 was not significant ($F=0.906$ and $p=0.650$), nor were the combined axes significant ($F=1.226$ and $p=0.110$).

Two species associated with burning in the 2001 season were *P. punctatissima* and *E. brigitta brigitta*. *A. oreas* was negatively associated with burning. Midlands Mistbelt grassland had one associated species, *A. taikosama*.

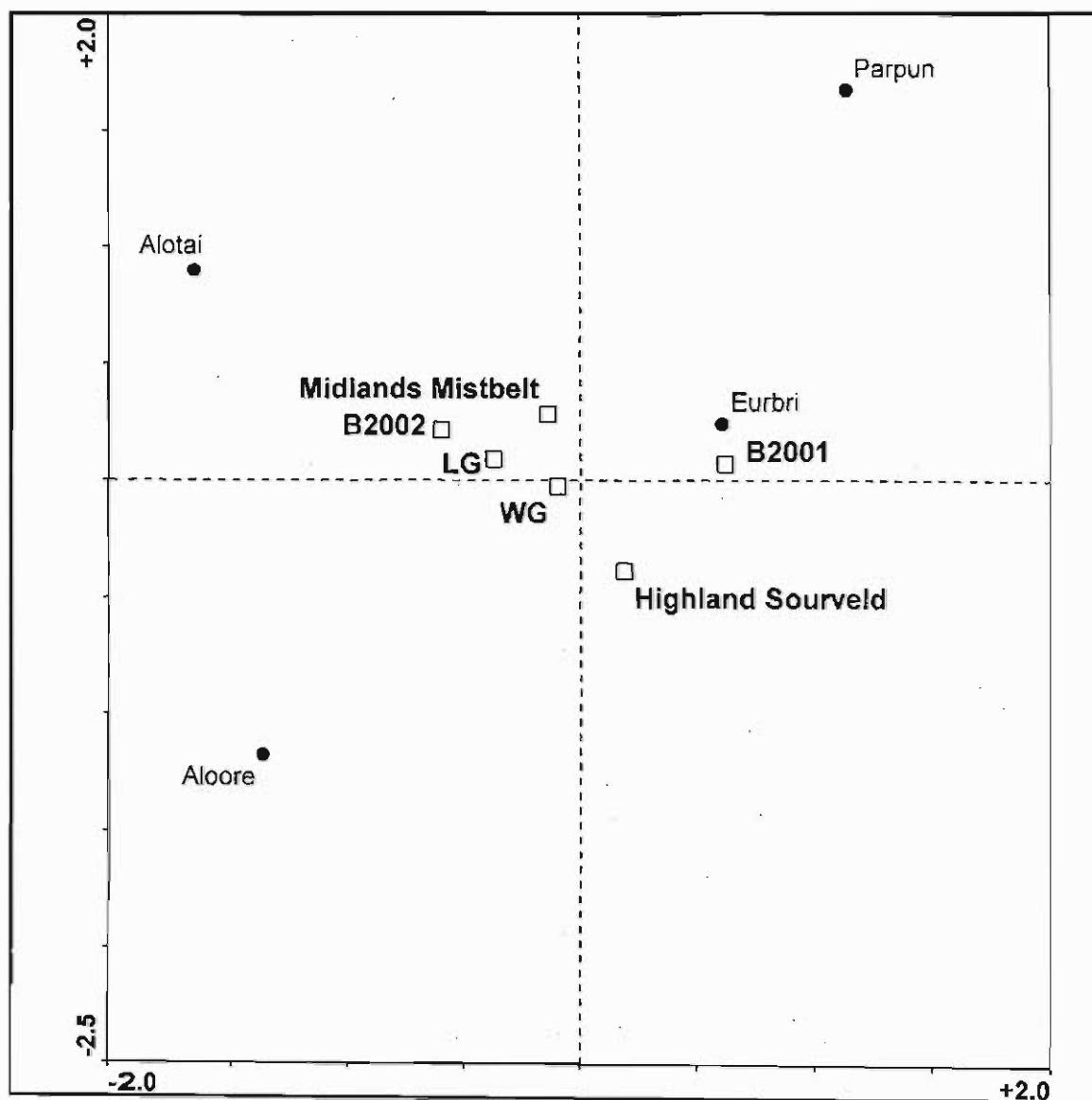


Figure 30. Canonical Correspondence Analysis (CCA) of butterfly species versus nominal variables. (Only species with >50% of their variance explained were plotted). The combined axes were not significant ($F= 1.226$ and $p= 0.110$) (LG= livestock grazed; WG= wildlife grazed; B2001= burnt in 2001; B2002= burnt in 2002). (Refer to Table 7 for butterfly species codes).

3.11. Grasshopper feeding guilds

Analysis of variance was done to test whether there was a significant difference between the feeding guild structure of the grasshopper assemblage and the type of grassland that the assemblage occupied. No significant difference between the feeding guilds was found when the two grassland types, Highland Sourveld and Midlands Mistbelt were compared (Table. 21).

Table 21. A one-way ANOVA of feeding guilds versus Veld Type (Highland Sourveld and Midlands Mistbelt) for grasshopper species across the ten study sites

Feeding guilds	df	F-statistic	p-value
Ambivorous	1	3.47	0.10
Forbivorous	1	0.92	0.36
Graminicolous	1	0.36	0.56
Grass eaters	1	1.43	0.27

3.12. Assemblage responses to grazing regimes

3.12.1. Grasshoppers

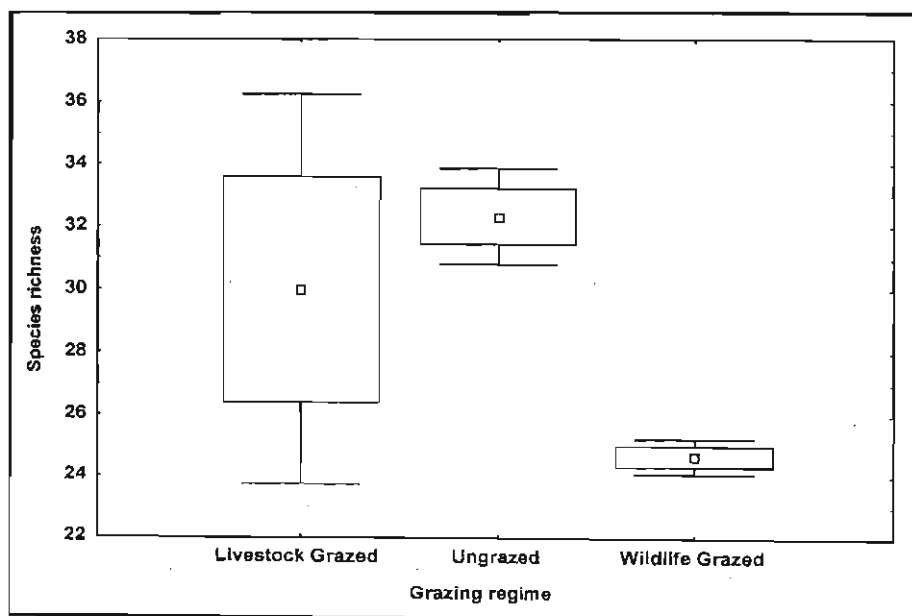


Figure 31. A box and whisker plot diagram comparing species richness and type of grazing (LG= livestock, UG= Ungrazed and WG= wildlife grazed).

Table 22. A one-way ANOVA of species richness and species evenness versus land-use type (Grazing) for grasshopper species across the ten study sites

Grasshoppers	df	F-statistic	p-value
Evenness	2	1.10	0.39
Richness	2	3.34	0.11

Table 23. The LSD post hoc test was undertaken to show any significant differences between grazing strategies and species richness (5% significance denoted by *) (LG= livestock, UG= Ungrazed and WG= wildlife grazed)

Grazing	LG	UG	WG
Mean richness	30	32.3	24.67
LG	----	0.47	0.13
UG	0.47	----	0.05*
WG	0.13	0.05*	----

Wildlife grazed sites were significantly different in terms of species richness from ungrazed sites at the 5% level (Table 23). The species richness means were 32.3 and 24.7 for ungrazed and wildlife grazed lands respectively (Fig. 31 and Table 23).

3.12.2. Butterflies

No significant difference was found when comparing butterfly species richness across the three different grazing regimes at the ten sites. The same could be said for abundance with no significant variation across the different grazing regimes (Table 24).

No significant results were achieved when undertaking a post hoc LSD test, for species richness across the various grazing types.

Table 24. Butterfly richness and abundance when compared across the three different grazing types

Sample	df	F-statistic	p-value
Butterfly richness	2	1.45	0.30
Butterfly abundance	2	0.05	0.95

3.13. Grasshopper responses to forestry

During the months of February and March the forestry compartments that adjoined Goodhope 1 were sampled for grasshopper species. The sampling was concluded after the findings for the two months indicated that, forestry did not provide suitable habitat for grasshoppers, especially Acridoidea that were totally absent. Table 25 represents the findings and comparisons between plantations and open grassland.

Table 25. The abundance of grasshoppers when comparing forestry (plantations) with the indigenous grassland. Figures in parentheses indicate the number of species recorded

Month	Forestry	Grassland
February	6 (2)	188 (11)
March	22 (2)	466 (19)

CHAPTER 4

INDICATORS OF GRASSLAND CONDITIONS

4.1. Surrogate subsets of grasshoppers

The surrogate subset comparisons were undertaken in order to test whether or not family richness and endemic richness in this study would be representative of the total species richness, in order to facilitate the use of families or endemics as surrogates for the entire species richness.

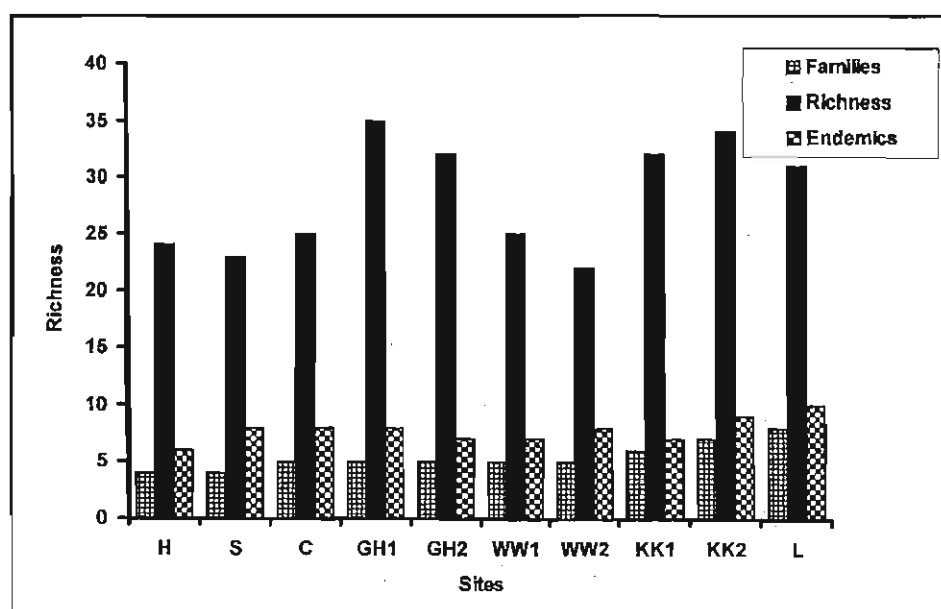


Figure 32. Total grasshopper species richness plotted against family richness and number of endemics for the ten study sites. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2).

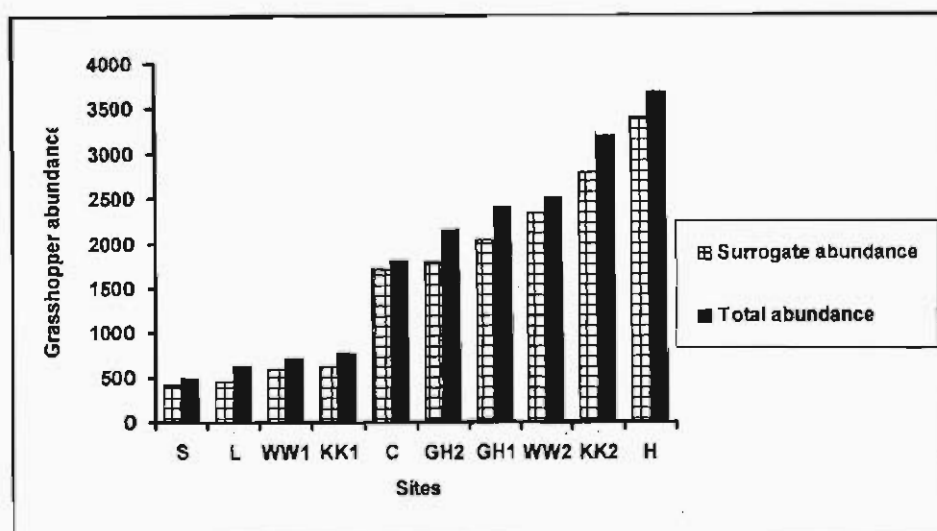


Figure 33. Surrogate grasshopper abundance for the ten sites compared to total grasshopper abundance. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2).

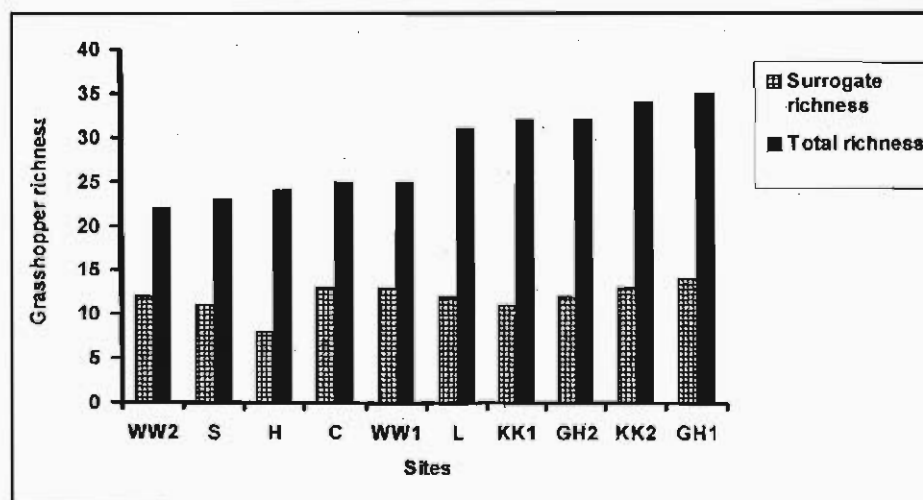


Figure 34. Surrogate grasshopper richness for the ten sites compared to total grasshopper richness. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahrenonga 1, WW2= Wahrenonga 2).

4.2. Surrogate subsets for butterflies

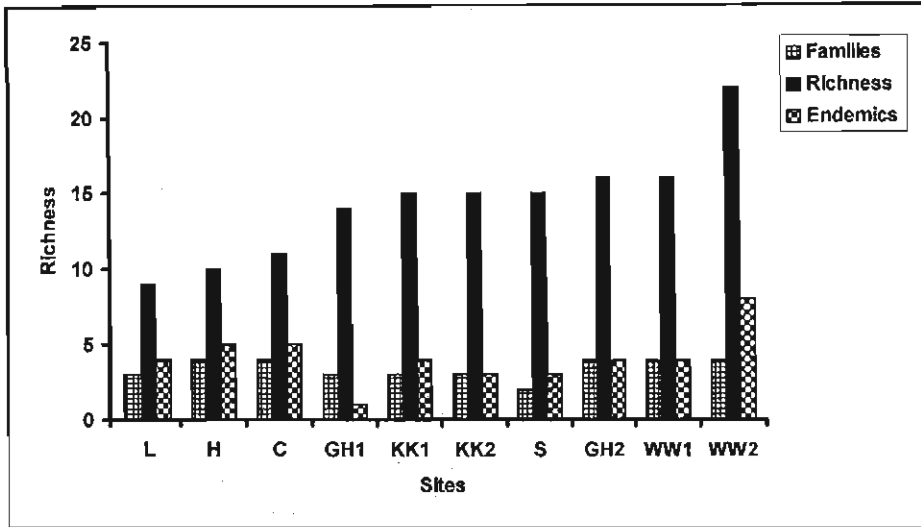


Figure 35. Total butterfly species richness plotted against family richness and endemics for the ten study sites. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).

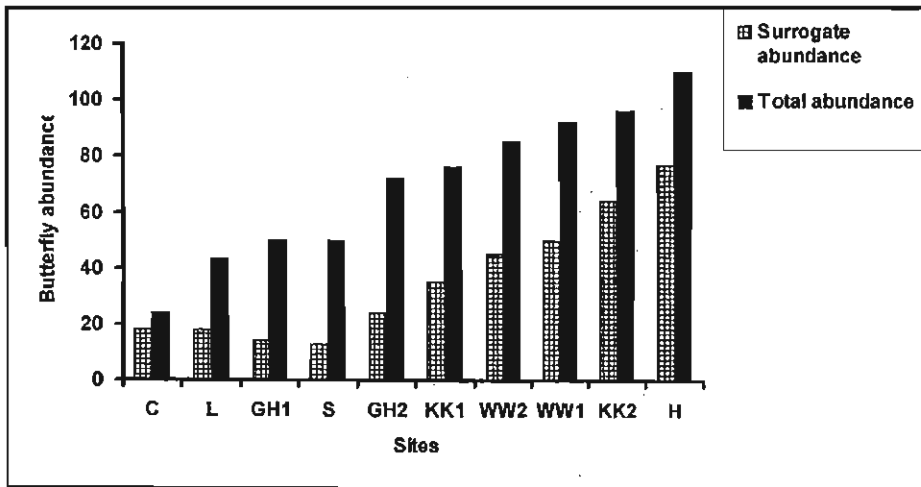


Figure 36. Surrogate butterfly abundance for the ten sites compared to total butterfly abundance. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).

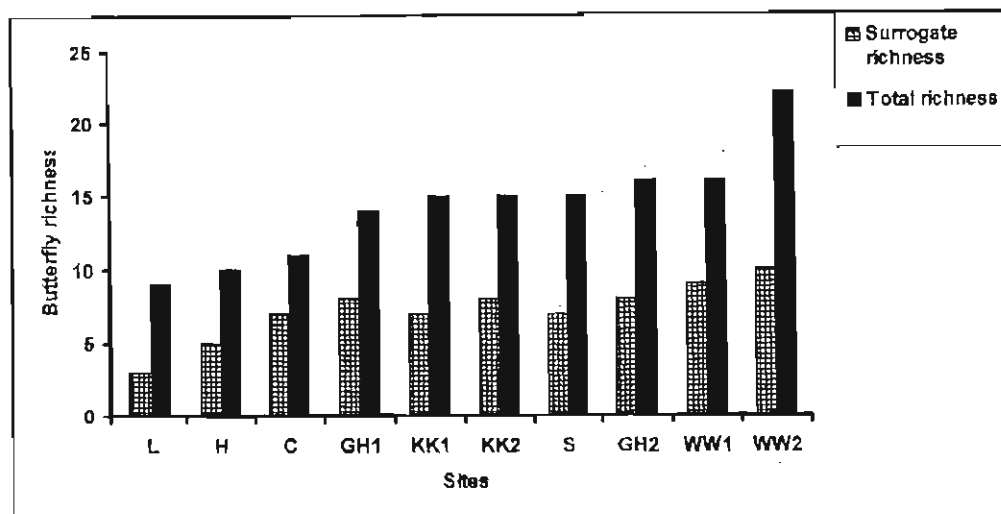


Figure 37. Surrogate butterfly richness for the ten sites compared to total butterfly richness. (C= Coleford, GH1= Goodhope 1, GH2= Goodhope 2, H= Himeville, KK1= Karkloof 1, KK2= Karkloof 2, L= Linwood, S= Stirling, WW1= Wahroonga 1, WW2= Wahroonga 2).

4.3. Correlations between grasshoppers, butterflies & grasses

When comparing species richness across the three sample taxa a significant correlation was apparent between grasshopper species richness and grass species richness (Table 26). There was a negative correlation between grasshopper species richness and butterfly species richness. Butterfly and grass species richness showed no correlation. When comparing endemic grasshopper and butterfly richness across the sites there was a negative correlation. No correlations were done for endemic grasses and butterflies and grasshoppers as only one species of grass encountered was exotic (only recorded at two sites) and endemism richness would closely mirror total species richness.

At the family richness level no correlation was found between grasshoppers and grasses, as at the species level. When comparing grasshoppers to butterflies and grasses to butterflies they were negatively correlated, but not significant (Table 26).

Table 26. Pearson's correlation values when comparing like categories between the butterfly and grasshopper assemblage. * = Significant at the 0.01 level

Categories	r-value	df	p-value
Family richness			
Butterflies vs. Grasses	-0.06	8	0.87
Grasshoppers vs. Grasses	0.21	8	0.57
Grasshoppers vs. Butterflies	-0.20	8	0.58
Species richness			
Butterflies vs. Grasses	0.01	8	0.98
Grasshoppers vs. Grasses	0.75	8	0.01*
Grasshoppers vs. Butterflies	-0.14	8	0.68
Endemic richness			
Grasshoppers vs. Butterflies	-0.15	8	0.67

4.4. Grasshopper surrogate subset

A Mantel's test was undertaken to determine whether the surrogate grasshopper species chosen would reflect the overall grasshopper species richness as well as total abundance encountered over the twelve-month study period. In both instances the two matrices were significantly correlated, with a higher correlation for the abundance data (Table 27). 1000 simulations were undertaken for each of the two correlation matrices with all runs recording an average Z less than the observed Z.

Table 27. Mantel's test correlations between total and surrogate grasshopper species richness, and total surrogate species abundance. Observed Z greater than average Z from the randomized runs indicates a positive association

Data matrices	Number of samples	observed Z	average Z	Mantel's r statistic	p-value
Species richness	116	0.69	0.63	0.90	0.001***
Species abundance	116	0.95	0.89	0.99	0.001***

In addition to the Mantel's r statistic being significant, the numbers of samples needed to obtain a 95% confidence interval (C.I.) were calculated for the full data set and the surrogate data set (Table 28).

Table 28. The number of samples (transects that need to be walked) needed to achieve the relevant confidence intervals (C.I.), for total species assemblage and surrogate grasshopper species assemblage

Confidence Intervals	Surrogate species assemblage (transects)	Total species assemblage (transects)
50%	3	7
75%	7	22
90%	14	51
95%	21	70
100%	70	118

In order to obtain 95% of the species richness for the total species assemblage, 70 transects would need to be walked to achieve the desired sampling efficiency as opposed to 21 transects for the surrogate species assemblage (Table 28).

4.5. Grasshopper assemblage responses to environmental categories

Nine categories were selected for, to compare compositional differences in grasshopper abundance and frequency of occurrence across the ten study sites. All nine categories were subdivided into two classes. These classes represented divisions in forb percentage (HIGH/LOW), Vegetation Height (TALL/SHORT), Aspect (COOL/WARM), Veld condition (GOOD/POOR), Grazing (GRAZED/NOT GRAZED), Biennial Burning (BIENNIAL/OTHER), Grassland type (MIDLANDS MISTBELT/HIGHLAND SOURVELD), Disturbance (HIGH/LOW) and Season in which the grassland was burnt (SPRING/WINTER).

Of the nine categories tested using Multiple Response Permutation Procedures (MRPP) three categories were not significant $p > 0.05$, one was significant at $p = 0.05$ and four were highly significant at $p = 0.01$ (Table 29). The more negative the T-statistic the greater the separation between the two categories tested.

Table 29. The results of Multiple Response Permutation Procedures (MRPP) for nine categories using the grasshopper abundance data collected across the ten sites for the twelve-month study period. (n.s. = Not significant, * = Significant $p = 0.05$, ** = Significant $p = 0.01$ and * = Significant $p < 0.01$)**

Category	Test Statistic (T)	p - value	Significance
ASPECT	-3.64	0.01	**
FORB CONTENT	-3.38	0.01	**
VEGETATION HEIGHT	-14.15	0.00	***
GRASSLAND TYPE	-1.70	0.07	n.s.
VELD CONDITION	-9.09	0.00	***
GRAZING	-2.02	0.05	*
BIENNIAL BURN	-0.77	0.18	n.s.
DISTURBANCE	-1.66	0.07	n.s.
SEASON OF BURN	-2.29	0.04	*

4.6. Grasshopper indicator species

33 species of grasshopper were significantly different when comparing categories of topography, vegetation structure, architecture and management. The species are separated according to their categories and detailed in the subsequent pages.

4.6.1. Topographic indicators

1. Aspect

The various aspects were combined into two classes. The south facing slopes were grouped together to represent cool slopes, with the remaining aspects grouped to represent warm slopes. Eight species were indicative of aspect, with seven species preferring cool slopes. The species were, *Acorypha nigrovariegata tibialis*, *Aneuryphymus montanus*, *Cyrtacanthacris aeruginosa*, *Lentula minuta*, *L. obtusifrons* and *Oxya hyla hyla* (Table 30).

One species, *Qachasia fastigiata* preferred warm slopes, i.e. north, east or west facing slopes. Appendix 1. Table 1. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

Table 30. Eight species were indicative of slope orientation (Aspect). Aspect = Warm or Cool Slopes. (Warm = -, Cool = +). Refer to Table 6 for grasshopper species codes

<u>Spp. Code</u>	<u>Aspect</u>
ACOTIB	+
ANEMON	+
CYRAER	+
GYMTEM	+
LENMIN	+
LENOBT	+
OXYHYL	+
QACFAS	-

4.6.2. Grasshoppers indicative of four vegetation categories

21 species responded to the separation of four vegetative environmental variables measured during sampling.

1. Proportion of forbaceous plant material

The percentage of forbaceous plant material was split into two categories High and Low. The High categories recorded greater than or equal to 22% of the vegetative material at each of the sites. Five sites were classified as High and five sites were classified as Low. The Low forbaceous sites had a low range of values falling between 14 and 17% percent. Ten species were representative of these two divisions.

Three species, *Lentula minuta*, *Heteropternis guttifera* and *Eyreprocnemis plorans* were positively indicative of grasslands that had a High forbaceous content, while six species were indicative of grasslands that had Low forbaceous content. The species were *Acorypha nigrovariegata tibialis*, *Calliptimicus semiroseus*, *Cruciotacris cruciata*, *Eremidium basuto*, *Gastrimargus drakensbergensis* and *Ochrophlebia caffra* (Table 31). Appendix 1. Table 2. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

2. Vegetation height

Vegetation height was separated into two distinct categories. Seven sites were representative of short grasslands, i.e. the vegetation was 65 cm or less. Eight species of grasshopper were representative of these two classes (Table 31).

The tall grasslands were represented by three sites with an average vegetation height of greater than 90cm and less than 120cm. Six species were indicative of short grassland, namely, *Acorypha nigrovariegata tibialis*, *Cruciotacris cruciata*, *Gastrimargus drakensbergensis*, *Ochrophlebia caffra*, *Orthochtha dasyncnemis* and *O.rosacea*.

Two species were indicative of tall grasslands, *Heteropternis guttifera* and *Lentula minuta*. Appendix 1. Table 3. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

3. Grasshopper species representative of grassland veld condition

The Key Veld Condition scores were split into two categories, Poor condition and Good condition. Six sites represented Good condition, with the remaining four sites representing Poor condition. 12 species of grasshopper were representative of these two classes. Seven species were indicative of Good veld condition. The species were, *Acorypha nigrovariegata tibialis*, *Calliptimicus semiroseus*, *Cruciotacris cruciata*, *Gastrimargus drakensbergensis*, *Ochrophlebia caffra*, *Orthochtha dasyncnemis* and *O.rosacea*.

Five species of grasshopper represented Poor veld condition scores. The species were, *Cyrtacanthacris aeruginosa*, *Heteropternis guttifera*, *Lentula minuta*, *L. obtusifrons* and *Oxya hyla hyla* (Table 31). Appendix 1. Table 4. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

4. Grassland type

The grasslands were split into two categories based on the grassland type described by Camp (1998). The two categories were Highland Sourveld and Midlands Mistbelt. 13 species of grasshopper were indicative of the two grassland divisions (Table 31).

Acorypha nigrovariegata tibialis, *Acrida acuminata*, *Catantops melanostictus*, *Erimidium basuto* and *Ochrophlebia caffra* were indicative of Highland Sourveld grasslands. Eight species of grasshopper were indicative of Midlands Mistbelt grasslands. The species were, *Anthermus granosus*, *Conocephalus* sp., *Faureia milanjica*, *Lentula minuta*, *Orthochtha dasycnemis*, *Oxya hyla hyla*, *Rhaphotittha cephalica* and *Whitea alticeps*. Appendix 1. Table 5. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

Table 31. 21 species were indicative of various vegetative assessments. Forbcon= % forbaceous material (Low = +, High = -), Veg height = vegetation height (cm) (Short = +, Tall = -), VCKey= Veld Condition scores (Poor = -, Good= +) and Grasstype = Grassland Type (Highland Sourveld or Midlands Mistbelt) (Highland Sourveld = +, Midlands Mistbelt = -). ++, -- denotes a very strong indicator. Refer to Table 6 for grasshopper species codes

Spp. code	Forbcon	Veg height	VCKey	Grasstype
ACOTIB	+	+	+	+
ACRACC	0	0	0	+
ANTGRA	0	0	0	-
CALSEM	+	0	+	0
CATMEL	0	0	0	+
CONSPP	0	0	0	-
CRUCRU	++	++	++	0
CYRAER	0	0	-	0
EREBAS	+	0	0	+
EYPPLO	-	0	0	-
FAUMIL	0	0	0	-
GASDRA	+	+	+	0
HETGUT	-	-	-	0
LENMIN	-	0	-	-
LENOBT	0	0	-	0
OCHCAF	+	+	+	+
ORTDAS	0	++	+	--
ORTROS	0	+	+	0
OXYHYL	0	0	-	-
RHACEP	0	0	0	-
WHIALT	0	0	0	-

4.6.3. Management Indicators

24 species of grasshopper were indicative of the various management regimes encountered through the study period. The four testable regimes were, grazing, biennial burning of the grassland patches, whether the grassland were over-utilised (Disturbance) and the season in which they were burnt (Table 32).

1. Grazing

Four grasshopper species, *Acorypha nigrovariegata tibialis*, *Acrida acuminata*, *Gymnobothrus temporalis temporalis* and *Ochrophlebia caffra* were positively associated with grazing, while five species were indicative of rested or not grazed grassland (Table 32). The species were *Catantops melanostictus*, *Conocephalus* sp., GRYLLIDAE, *Ornithacris cyanae* and *Qachasia fastigiata*. Appendix 1. Table 6. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

2. Biennially burnt grasslands

Six species of grasshopper were positively and negatively associated with the periodicity of burning. Only one species responded positively to biennial burning, *Phloeonotus humilis*.

Five species of grasshopper preferred the grassland to be burnt triennially or not at all. *Dirshia abbreviata*, *Eyprepocnemis plorans*, *Heteropternis guttifera*, *Qachasia fastigiata* and *Whitea alticeps* (Table 32). Appendix 1. Table 7. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

3. Disturbed grasslands

Five species of grasshopper were indicative of the amount of disturbance of grasslands. Three species were positively associated with disturbed grasslands, *Acrida acuminata*, *Calliptamulus natalensis* and *Calliptimicus semiroseus*. *Orthochtha dasyncnemis* and *Qachasia fastigiata* were positively associated with grasslands where there was no or limited disturbance (Table 32). Appendix 1. Table 8. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

4. Season in which the grasslands were burnt

Nine species of grasshopper indicated a preference for the season in which grasslands were burnt (Table 32). Only one species was positively correlated with spring burning, *Acrida acuminata*. The remaining eight species preferred grasslands that were burnt during the winter period. The species were *Anthermus granosus*, *Cyrtacanthacris aeruginosa*, *Faureia milanjica*, *Gastrimargus crassicollis*, *G. determinatus vitripennis*, *Omithacris cyanae*, *Phaeocatantops sulphurius* and *Rhaphotittha cephalica*. Appendix 1. Table 9. contains the results for all species tested using Indicator Species Analysis (I.S.A.).

Table 32. 24 species were indicative of various management regimes. Grazing = Livestock or Wildlife grazed (Grazing = +, Not grazed = -), Biennial = Biennially burnt (Biennial = -, Other = +), Disturbed = Disturbance (Over-utilisation) (Low = -, High = +) and Season of burn = winter or spring (winter = -, spring = +). -- denotes very strong indicator. Refer to Table 6 for grasshopper species codes

Spp. Code	Grazing	Biennial	Disturbed	Season of burn
ACOTIB	+	0	0	0
ACRACC	+	0	+	+
ANTGRA	0	0	0	-
CALNAT	0	0	+	0
CALSEM	0	0	+	0
CATMEL	-	0	0	0
CONSP	-	0	0	0
CYRAER	0	0	0	-
DIRABB	0	+	0	0
EYPPLO	0	+	0	0
FAUMIL	0	0	0	-
GASCRA	0	0	0	-
GASVIT	0	0	0	-
GRYLL	-	0	0	0
GYMTEM	+	0	0	0
HETGUT	0	+	0	0
OCHCAF	+	0	0	0
ORNCYA	-	0	0	-
ORTDAS	0	0	--	0
PHASUL	0	0	0	-
PHLHUM	0	-	0	0
QACFAS	-	+	-	0
RHACEP	0	0	0	-
WHIALT	0	+	0	0

4.7. Butterfly assemblage differences

The categories utilised for butterfly compositional difference were the same as the categories used when comparing grasshoppers. Of the nine categories tested using MRPP only one (Season of burn) was significant at $p = 0.05$ (Table 33) the remainder were not significant.

Table 33. The results of Multiple Response Permutation Procedures (MRPP) for nine categories using the butterfly abundance data collected across the ten sites for the twelve-month study period. (n.s. = Not significant and * = Significant $p = 0.05$)

Category	Test Statistic (T)	p - value	Significance
Aspect	1.22	0.92	n.s.
Forb content	-0.18	0.38	n.s.
Vegetation height	-0.30	0.33	n.s.
Grassland type	-0.48	0.28	n.s.
Veld condition	0.39	0.60	n.s.
Grazing	-0.26	0.34	n.s.
Biennial burn	-1.57	0.07	n.s.
Disturbance	0.50	0.64	n.s.
Season of burn	-2.56	0.02	*

4.7.1. Topographic indicators

1. Aspect

Only one butterfly species, *Harpenderyeus noquasa* was associated with aspect. This species was indicative of cool slopes, i.e. southerly slopes.

Table 34. Eight species were indicative of slope orientation (Aspect). ASPECT = Warm or Cool Slopes. (WARM = -, COOL = +). Table 7 for butterfly species codes

Spp. code	Aspect
HARNOQ	+

4.7.2. Vegetative indicators

Three species were significant when comparing vegetative characteristics. *Belenois zochalia zochalia* was indicative of short grassland with a low veld condition score. *Colias electo* and *Papilio demodocus* were indicative of high and low forbaceous plant content respectively (Table 35).

Table 35. Three butterfly species were indicative of various vegetative assessments. Forbcon= % forbaceous material (Low = +, High = -), Veg height = vegetation height (cm) (Short = +, Tall = -), VCKey= Veld Condition scores (Poor = -, Good= +) and Grasstype = Grassland Type (Highland Sourveld or Midlands Mistbelt) (Highland Sourveld = +, Midlands Mistbelt = -). Table 7 for butterfly species codes

Spp. Code	Forbcon	Veg height	VCKey	Grasstype
COLELE	-	0	0	0
PAPDEM	+	0	0	0
BELZOC	0	+	-	0

4.7.3. Management indicators

In terms of management, butterfly species were more prone to be indicative of various management regimes, as opposed to vegetative characteristics and topography. Ten species were indicative of the four management categories.

1. Grazing

Two species, *Actizera lucida* and *Pardopsis punctatissima punctissima* were indicative of grasslands that were not exposed to any form of grazing (Table 36).

2. Disturbance

Vanessa cardui and *Catopsilia florella* were indicative of highly disturbed grassland patches. Their abundance was highest at these patches, and in the case of *V. cardui* it was the most abundant butterfly species. (Table 36).

3. Biennially burnt grasslands

Five species of butterfly were indicative of grasslands that were not exposed to fire or were triennially burnt (Table 36).

4. Season in which the grassland was burnt

Four species of grasshopper were indicative of the season in which the grasslands were burnt. Three species were indicative of grasslands that were burnt in winter, and one species was indicative of grasslands burnt in spring (Table 36).

Table 36. 10 species were indicative of various management regimes. Grazing = Livestock or Wildlife grazed (Grazing = +, Not grazed = -), Biennial = Biennially burnt (Biennial = -, Other = +), Disturbed = Disturbance (Over-utilisation) (Low = -, High = +) and Season of burn = winter or spring (winter = -, spring = +). Table 7 for butterfly species codes

Spp. Code	Grazing	Disturbance	Biennial	Season of burn
ACTLUC	-	0	0	-
COLELE	0	0	0	-
PARPUN	-	0	0	-
ALOORE	0	0	+	0
BELAUR	0	0	+	0
ORALAC	0	0	+	0
PAPDEM	0	0	+	0
PSEVAR	0	0	+	+
CATFLO	0	+	0	0
VANCAR	0	+	0	0

CHAPTER 5

DISCUSSION

5.1. Grassland types

Spatial scale is of paramount importance when the focus of the study is invertebrate assemblages and the development of patterns across grasslands (Gaston *et al.*, 1993, Palmer & White, 1994; Thomas & Abery, 1995). The behaviours of organisms are influenced by the scale at which the landscape pattern is perceived (Kotliar & Wiens, 1990; With, 1994). The finest resolution of scale that is perceived by an organism ("grain") is contrasted with the broadest perceptual resolution ("extent") (McIntyre & Wiens, 1999). Weather also has a major impact on grasshopper species (Kindvall, 1996) and assemblages (Gandar, 1982a; Rodell, 1997) both directly and on the plants on which the insects depend.

Yoshino (1965) has categorized spatial scale into three major units:

1. **Microscale-** the smallest unit, plant size both vertically and horizontally, plant architecture and the area directly surrounding the plant. This would be the microhabitat in which invertebrates live.

2. **Mesoscale-** this is a medium sized region, for example, the grassland area as a complete entity, with movement being undertaken over several hundreds of metres. The invertebrates may utilise this whole entity, or part thereof depending on specific requirements during their lifecycles. Uvarov (1977) showed that local migrations occur within grasslands from hatching through the various instars of development in grasshopper species as requirements change. Only a proportion of individuals become involved in these movements, while return movement to breeding habitats are not pronounced (Stebaev, 1970). When grasshoppers need to oviposit they require areas of bareground that have high light intensity and equivalent heat units (Onsager, 1963; Anderson & Hastings, 1966, Uvarov, 1977). Feeding grasshoppers require a well-vegetated area, because food resources and protection are in close

proximity. Local topography, disturbance and management of the grasslands determine grasshopper assemblages. These factors combine to determine the mesoscale and influence grasshopper assemblages.

3. Macroscale- this is the largest unit of scale, constituting a massive area, for example, a biome or a country that show similar traits throughout (Brown, 1995). In this study the macroscale are the two grassland types, Highland Sourveld and Midlands Mistbelt. Camp (1999) refers to these grasslands as Bioresource Units, distinguishable by abiotic and biotic variables, such as grass composition, soil type, rainfall and elevation.

Species diversity, therefore, at any one point in the landscape is determined by multiple factors acting at multiple scales (Turner, 1989; Turner & Gardner, 1991; Wiens, 1989; Debinski *et al.*, 2001).

One of the important questions that requires answering is, 'Can this distinction that is being made at the grassland type, macroscale level (separation into Midlands Mistbelt and Highland Sourveld for example) be applied to invertebrate taxa living within the grasslands?' In this study, no significant differences in grass and grasshopper species richness and abundance were found when comparing grassland type (Combining of the five Highland Sourveld grasslands and the five Midlands Mistbelt grasslands into two separate categories). This suggests that the climatic factors and 'grassland type' were relatively uniform for the ten different sites.

Differences in species composition, richness and abundance however are significant when assessing the grasslands as single entities and comparing them. In this study, it was possible to compare individual sites, as the grassland type in terms of grasshopper assemblages at the macroscale are not significantly different and any significant variation is due to the meso- and micro-scale.

5.2. Impact of abiotic variables on grasshopper assemblages

5.2.1. Slope orientation

Slope orientation can have a significant affect on the local distribution of grasshopper species. The thermal environment is extremely important to grasshoppers because temperature affects all biological functions and ultimately fitness (Willott & Hassall, 1998; Pitt, 1999). Two sites in this study, one north facing, the other east facing, showed significantly higher species richness, which mirrored what Samways (1990) found in the Natal Drakensberg. Grasshoppers orientate themselves in this manner to gain a thermal advantage (Anderson *et al.*, 1979; Chappell, 1983; Lactin & Johnson, 1997, 1998; Chappell & Whitman, 1990) as insolation is affected dramatically by aspect, with the limiting constraints on the grasshopper assemblage being light intensity, heat units and food (Uvarov, 1977; Porter & Redak 1996). Insolation can lead to substantial thermal differences between slope faces. Chappell (1983) demonstrated that overheating in montane and alpine species of grasshopper is essentially impossible, and the grasshoppers consistently expose their bodies to direct sunlight for the optimal body temperature required for their life activities. In semi-arid areas like the Karoo grasshoppers have a different behaviour, where equivalent time is spent basking and sheltering from heat as they are generally exposed to extremely high mean temperatures (Gebeyehu & Samways, 2002).

Fry and Lonsdale (1991) found that in the United Kingdom, a northern hemisphere country, that slope aspect had a significant affect on species richness and the presence of rare species. In the northern hemisphere southerly slopes are equivalent to northly facing slopes in South Africa. Studies conducted on the wart-biter (*Decticus verrucivorus*) a member of the Tettigoniidae showed that slope orientation played a significant role in the rate of instar development (Ingrisch, 1986) and the fecundity of females (Haes *et al.*, 1990). The two sites, (Linwood and Karkloof 2) which had the greatest number of grasshopper endemics/rarities, were the warmer more sheltered slopes, confirming that local topography does have an affect on grasshopper assemblages.

5.2.2. Elevation

Elevation has a significant affect on species richness, which declines with increasing elevation (Mani, 1961, Claridge & Singhrao, 1978; Currie, 1991; Thomas, 1991; Kennedy, 1994; Wettstien & Schmid, 1999).

20% of the sites in this study did not follow the trend of decreasing species richness with increasing elevation. The first site was more heterogeneous in terms of grass composition than other sites and hence the higher species richness (Otte, 1976; Gebeyehu & Samways, 2002). The other site is at high elevation, north facing and just below the crest of a hill. Samways (1990) showed that hilltops act as thermal refugia for grasshoppers. It is probable that elevation and aspect have a strongly interactive relationship, adding further variation to grasshopper assemblages. The probable explanation for increased richness is cold air drainage, which takes place on still winter nights, where the cold air drains down into the valleys, leaving the warmer air to blanket the hilltops. Cold air drainage is basically the stratification of air at various levels, increasing in temperature from valley floor to hilltop. Tyson and Crimp (1998) showed major fluctuations in diurnal temperatures, ranging from 14°C in the summer and 20°C in the winter. Such a large temperature fluctuation is going to have an affect on grasshopper species, as they are small, ectothermic and require high heat units for survival (Chappell, 1983). This effect is magnified if the grassland is burnt (Samways, 1990) or over-utilised, as it removes vegetation that would comprise grasshopper microhabitats. At cool body temperatures (<15°C), grasshoppers move slowly and are unable to feed, whereas at high body temperatures (>45°C) enzymes denature and death occurs (Chappell & Whitman, 1990). The optimal field temperature for grasshoppers is much narrower ranging between 35-42°C (Dempster, 1963; Chappell & Whitman, 1990). Grasshopper abundance figures, did not show a similar trend, with elevation not appearing able to predict abundance levels, as they are more closely related to management practices (Onsager, 1993, 2000).

5.3. Impact of biotic variables on grasshopper assemblages

5.3.1. Vegetation structure and architecture

Vegetation structure appears to play an important role in the determination of grasshopper species richness and abundance. Within grassland areas, vegetation shows a gradation of variation in grass and forbaceous species (Tainton, 1984a; Acocks, 1988). Vegetation structure is determined by many factors, the three main factors being soil type, amount of water retained in the soil and light intensity.

Grass species demonstrate trade-offs between being tall and developing a large root system (Tilman, 1982). Taller species occurring in this study area directed the majority of their resources to the development of large leaf areas, with little development of their roots. Root development is secondary as the soils in which these particular species grow are nutrient rich with plants not requiring large root masses to accumulate water and nutrients for growth and development. The grasses also need to grow tall so as to compete for light with the taller forbaceous species that occur in these wetter areas. Grasses (*Themeda triandra*, *Tristachya leucothrix*) that grow on slopes where the soil is denuded of moisture and are shallower, direct the bulk of their resources to the development of expansive root systems that can accumulate the water and resources needed for growth and development (Tilman, 1982). These grass species tend to be more important in terms of nutritional value for livestock (van Oudtshoorn, 1992).

Trade-offs allow grasses to compete better with one another for nutrients, water and light. Light intensity plays a significant role in the prediction of grass species (Tilman, 1982). Where the taller species of grass occur, they outcompete the shorter species and hence patchiness develops within the grassland. Acocks (1988) found that grass species growing in deeper, wetter soils and areas of shading, were sweeter and had a higher nutritional value, in terms of livestock grazing. The grass compositions in these areas were usually tall grass species (>90cm). The species encountered were *Miscanthus capensis*, *Cymbopogon excavatus* and *Hyparrhenia hirta*. These three species are considered poor in terms of their grazing value, with the exception of *H. hirta*, which is of average value when regrowth appears after defoliation. The suitability of any area of grassland for any one species is dependent on two variables. Firstly, the

presence, absence or abundance of its food plant(s) and, secondly, vegetation structure.

Many authors have shown that vegetation height and structure has a limiting affect on grasshopper assemblages, by not allowing enough light and heat to penetrate deep into the sward where the grasshoppers are situated, thus defining their habitats (Anderson, 1964; Mulkern, 1967; Otte, 1976; Uvarov, 1977; Karieva, 1983; Evans, 1988a,b; Kemp *et al.*, 1990; Quinn *et al.*, 1991; van Wingerden *et al.*, 1991a; Fielding & Brusven, 1993a; Kisbenedek, 1995; Thomas & Marshall, 1999; Meek *et al.*, 2000). The taller grass height frequently resulted from the presence of species (e.g. *Cymbopogon excavatus*, *Hyparrhenia hirta* and *Miscanthus capensis*), which harbour fewer grasshoppers than the lower, more even sward (Gandar, 1983, Chambers and Samways, 1998). Tall, dense vegetation often implies more shelter (Bossenbroek *et al.*, 1977), more food (Gandar, 1983) and greater microclimatic stability (Duffey *et al.*, 1974; Samways & Moore, 1991). There is however likely to be more shading and lower temperatures within areas of tall dense vegetation (Anderson, 1964; van Wingerden *et al.*, 1999b). Grasshoppers are ectotherms, requiring light and heat for survival and are unable to benefit from the advantages associated with increased vegetation height. Samways and Moore (1991), working in the same geographical area as this study, found that grasshoppers showed a definite preference for areas that were sunlit. Joern (1982) considers temperature to be a more important factor than light in habitat selection, although here there was a definite preference for sunlit areas. This may be due to tolerances in some environmental changes so as to preserve uniformity in others (e.g. direct sunlight). Optimal field temperature and RH (relative humidity) for grasshoppers are 35-42°C and 50-75% respectively (Dempster, 1963; Chappell & Whitman, 1990), and periods during which this is the case, grasshopper abundances are much higher. Consequently, irradiance has a very pronounced influence in habitat selection by grasshoppers. The taller the vegetation the more shading effect occurs, reducing the number of sites where grasshopper species can oviposit and bask, thus reducing abundance and richness. Grasshoppers will therefore abandon these areas (Lockwood, 1993) and make small migrations to areas better suited to their requirements.

This study concurs with these findings, showing that areas where the grass sward is less than 45cm had the highest species richness and abundance, with the exception of Wahroonga 2 where the richness of grasshopper species was the lowest of all sites studied. It could be postulated that the reason for such low species richness, was that this particular site was regularly mowed and never burnt, affecting the natural process, which in turn molds the grasshopper assemblage.

It can therefore be concluded that grasshopper biomass is highly reduced when vegetation height is greater than 90cm, supporting the findings of other authors that plant physiognomy, rather than composition (Gandar, 1982a), play an important role in the formation and prediction of grasshopper assemblages.

5.3.2. Habitat heterogeneity and grasshoppers

The term heterogeneity can have many meanings (Kolasa & Pickett, 1991), but the relevant parameters in the current study derive from variability in vegetation structure, composition, density and biomass. This type of heterogeneity influences species diversity, the variety of habitats and ecosystem functioning (Christensen, 1997; Wiens, 1997; Bailey *et al.*, 1998; Fuhlendorf & Engle, 2001). Environments are heterogeneous in space and time (McIntyre & Wiens, 1999). In this study patterning results from a number of contributing factors, namely, slope, aspect, management geology and elevation. Patterning therefore is more natural than uniform of the grasslands that were sampled. The patterning of this heterogeneity affects the abundance and distribution of organisms and the array of population, community, and ecosystem patterns that follow from distribution and abundance (Robinson *et al.*, 1992). Therefore, heterogeneity is the precursor to diversity at most levels of ecological functioning and should serve as the foundation for ecosystem and conservation management (Christensen, 1997; Wiens, 1997). Maintenance of a heterogeneous landscape, where a variety of habitats is conserved, is essential for the full array of local invertebrate species, from bumblebees (Kells & Goulson, 2003) to dragonflies (Steytler & Samways, 1995). British bumblebees need a variety of field and forest boundary types, while South African dragonflies need a variety of lakeside vegetational structural types. Heterogeneity also impacts upon grasshopper assemblages. Heterogeneity depends on scale, with the relationship between ecological productivity and species diversity

changing with spatial scale (Chase & Leibold, 2002). Grassland may appear heterogeneous to humans as the scale at which we view the grassland is at a much lower resolution. Humans define grassland heterogeneity along the lines of plant species diversity, proportion of grasses to forbaceous plants as well as the proportion of vegetation in relation to bare ground (Fuhlendorf & Engle, 2001; Gutzwiller, 2002). For small invertebrates i.e. grasshoppers landscape is viewed at a much higher resolution, with changes in plant architecture and structure being more influential on the behaviour of individuals and species (McIntyre & Wiens, 1999).

In general, the more diverse the mix of plant species surviving in a community, the greater the potential to support a diversity of specialist invertebrates dependent on specific plant hosts (MacArthur, 1972; Crawley, 1983; Mullen *et al.*, 2003) and resultant creation of a more diverse array of habitats (Mulkern, 1967; Skinner, 2000; Beckerman, 2000). Landsberg *et al.*, (1999) showed that continual utilisation reduces floral diversity and impacts the indigenous biota thus reducing habitat heterogeneity. Ants, bees, Coleoptera and Lepidoptera are known to switch between habitats daily, while they undertake their daily routines of foraging, resting and reproduction (Duelli, 1997). The feeding habits and environmental requirements of exopterygotan insects are similar in the young and adult stages (Samways & Sergeev, 1997). Yet, the various life stages may be characterized by differing responses to landscape form and pattern at the various spatial scales. Invertebrates benefit from greater structural variety resulting from a fine scale mosaic of different sward heights (Fry & Lonsdale, 1991) as well as increased diversity amongst the plant species. Davidowitz and Rosenzweig (1998) showed that there is a positive correlation between habitat complexity and species diversity: the greater the heterogeneity of the habitat the greater the number of species in that habitat (Greatorex-Davies *et al.*, 1994; Mullen *et al.*, 2003; Weibull *et al.*, 2003).

In a mosaic landscape, a combination of varying biotopes and ecotones, the habitat heterogeneity is increased and all species appear to profit from this increased complexity. Increasing habitat heterogeneity reduces density fluctuations and extinction risk of local populations of the bush cricket *Metrioptera bicolor* (Kindvall, 1996). This has been substantiated by studies undertaken on butterflies, where local populations are more stable with increased habitat heterogeneity (Erhlich & Murphy, 1987; Weiss *et al.*, 1988; Swengel, 1996a). Within a grasshopper assemblage, individual species

have different environmental and management requirements as with other taxa. The problem of catering for these different requirements simultaneously to conserve maximum species richness and diversity can be solved with a system of rotational management (Chambers & Samways, 1998). Rotational management allows a wider spectrum of grasshopper species with respect to food and habitat requirements, through greater habitat heterogeneity. This however, can lead to conflict between the management of grasslands for floral diversity and management for diversity in other groups such as the invertebrates (Van Wieren, 1989; Grant *et al.*, 1996). It is rare, however, that all management objectives can be realized within the same area at the same time (Van Wieren, 1989).

The findings here are in accordance with this notion, as the site with the highest grasshopper species richness was Goodhope 1 with the greatest compositional difference in terms of grasses and microhabitats. Overall this grassland comprised three different habitats: 1) grassland, 2) grassland/wetland interface and 3) wetland. Grassland/wetland interface and the wetland habitats have a particular species assemblage, with high abundances of three species, namely, *Paracinema tricolor*, *Lentula obtusifrons*, and *Oxya hyla*. *O. hyla hyla* was only recorded at this site as it is adapted to live in marshy areas with expanses of open water, with the evolutionary development of flattened tibiae for swimming.

Karkloof 1 was also highly heterogeneous with two distinct layers within the grassland. Firstly, the woody plants (*Phymaspermum acerosum*) provide a unique habitat for the forbivorous species (*Lentula minuta*, *Cyrtacanthacris aeruginosa*, *Omithacris cyanae* and *Eupropracris cylindricollis*). Secondly, the small grass patches between the stands of tall woody species provide a microhabitat, for the graminicolous species, such as *Rhaphotittha cephalica* and *Gastrimargus vitripennis*. In addition to the complex plant architecture, this grassland is in a state of succession, from being a once forested area. Rehabilitation involves colonization by new species (Andersen, 1997), and the speed at which the colonization takes place with further successional development is a function of distance from the source population, vagility of the species and their method of dispersal. Andersen *et al.* (2003) showed that the rehabilitation of mine sites rarely reconstitutes the original fauna, with functional groups such as cryptic species, cold climate specialists and specialist predators generally underrepresented, while there is

a high relative abundance of dominant species, hot climate specialists and/or opportunists. Although the pioneer species were replaced by later successional species, the nature of the succession was patchy, with earlier successional remnant assemblages remaining. This may temporarily increase species richness. As the system stabilizes the species richness may also decline as habitat heterogeneity declines. However, should the grass/forb proportion remain static then the species richness could remain similar, but species turnover will take place. Nevertheless, we are still likely to see future catastrophic regime shifts in some ecosystems, where accumulation of pressures reach a point where an ecosystem changes from one state to another and then remains more or less static in the new state (Scheffer & Carpenter, 2003). This has been shown at sites in Germany, where butterfly species richness did not change during plant succession, although the species composition changed substantially (Steffan-Dewenter & Tschardtke, 1997). The aim of the custodians of this grassland is to remove all the *Phymaspermum acerosum*, returning it to natural *Themeda triandra* dominated grassland. With this as the desired end point and the management structured around achieving this, it will reduce the habitat heterogeneity and at least once species (*Eupropraxis cylindricollis*) that is currently only found at this site will disappear from the grasshopper assemblage.

To maintain species diversity it is therefore imperative that management of grasslands aims specifically for a heterogeneous landscape (Fuhlendorf & Engle, 2001) with management practices, particularly grazing, that varies from year to year (Onsager, 2000). Therefore, ranges of taxa need to be sampled and a decision made on which method supports the highest diversity across taxa.

5.4. Landscape level and the impacts associated with man

5.4.1. Defoliation

Grasslands in South Africa are not considered to be climax vegetation, and are dynamic. If left without management, they develop into scrub veld and eventually forest (Acocks, 1988; Hardy, 1999). Therefore an integral part of grassland maintenance is management. Management revolves around maintaining grasslands in their current state, (i.e. rich in short productive grass species) which rely heavily on defoliation through relatively frequent burning, mowing or grazing (Belsky 1992; Fuhlendorf &

Smeins 1997; Osem *et al.*, 2004). This form of defoliation results in the removal of litter, improving light availability and reducing soil nitrogen (Seastedt *et al.*, 1991; Ojima *et al.*, 1994) and above-ground net primary productivity (ANPP) (Tainton *et al.*, 1978; Milchunas & Lauenroth 1993; Snyman 2004).

Farmers and environmental agencies typically undertake the management of grasslands in South Africa. In the absence of this management important agricultural and environmental grass species are lost, as their vigour declines and they eventually die (Kruger, 1984; Tainton, 1984b). With lack of management, the grassland becomes patchy with large areas of bare ground, leading to soil erosion. Over-utilisation and repeated burning result in a loss of perennial decreaser species. This occurs through the loss of perennial grass species that play a key role in soil stabilization (O'Connor, 1996). Topsoil loss through erosion results in a species switch from perennial grasses to pioneer grasses and deeper-rooted shrubs in the long term (O'Connor, 1996; Dougill & Trodd, 1999; Manzano & Navar, 2000). Pioneer species along with shrubs are considered unpalatable by livestock leading to selective grazing, further degrading the grassland (van Oudtshoorn, 1992) and the development of further bare patches. Under-utilisation results in similar problems, as the grasslands become moribund, resulting in the grass tufts dying as light and rain infiltration are limiting factors. With the tufts dying out bare patches are opened up allowing for invasion by alien species and/or exposure of the soils to the elements and thus providing the potential for soil erosion to commence.

Several authors have demonstrated that a reduction in canopy height, and grass cover has an inverse relationship with grasshopper densities (Rivers-Moore & Samways, 1996; Samways & Lockwood, 1998; Lockwood *et al.*, 2000; Onsager, 2000). The grasshopper assemblage would constitute the common, more robust species of grasshopper with a reduction in rare and endemic species through microhabitat loss. These characteristics were mirrored in this study when the incorrect grazing strategies were implemented, a prime example being Stirling and Himeville where species richness was reduced through poor management.

1 Livestock grazing

Grazing is an important land-use of grasslands in the KwaZulu-Natal Midlands. Grazing is both beneficial and detrimental. The advantages are only applicable if management of grazing is well controlled and monitored, as many authors (e.g. Capinera & Sechrist, 1982; Dolek & Geyer, 1997; Zschokke *et al.*, 2000; Kruess & Tschardtke, 2002) have shown a decline in species richness and abundance. Changes in plant composition by livestock and wildlife are likely to have the greatest impact on host-specific herbivores, which are most commonly invertebrates (Crawley, 1983; Beckerman, 2000). Grazing by large herbivores removes some palatable grasses and forbs, while trampling of the soil affects its structure and vegetal composition (Taboda & Lavado, 1993). The trampling of soil and change (switch) in vegetation composition has dramatic effects on resident invertebrate fauna (Maliha *et al.*, 2000). Continuous grazing has reportedly had a deleterious effect on other taxa of soil-dwelling and above-ground invertebrates (King & Hutchinson, 1976). These effects are only noticeable when there is high intensity grazing on areas with poor resources. It is therefore pertinent at this time to provide insight into the types of grazing that occur in the study area.

At the sites where livestock grazing took place, the various managers have implemented three different grazing strategies. At the first site (Stirling) the grassland is continuously grazed, with typical patterns associated with continuous grazing such as patchiness and large areas of 'tall' unpalatable grass, such as *Miscanthus capensis* that is encroaching on small areas of remnant *Themeda triandra*. Coupled to the encroachment by *M. capensis* and *Aristida junciformis*, is the increased number of forbs and woody species that are now able to find a foothold as competition between themselves and the grasses has been reduced through continual trampling and grazing. The *M. capensis* and *A. junciformis* are beginning to outcompete the perennial *T. triandra*. Large areas of bareground are starting to form at this site through the continual use of paths by livestock, as well as by selective grazing and trampling. *M. capensis* is less able to deal with continual trampling, due to its growth form, which apportions a high percentage of available resources to leaf development, resulting in poor root mass growth, and subsequent soil destabilization. This is additive to the deleterious effects of continuous grazing. Bare patches are more frequently

encountered and open to erosion, notably by water, leading to a complete shift in vegetal species composition and also a decline in grasshopper richness.

Most grasshopper species avoid sites that are heavily grazed and trampled, and where there is less food (Holmes *et al.*, 1979; Dukas & Bernays, 2000), minimal shelter from natural predators (Belovsky & Slade, 1993), and high physical disturbance from hooves (Roux & Opperman, 1986, Bosch & Gauch, 1991; Rivers-Moore & Samways, 1996; Samways & Kreuzinger, 2001; Gebeyehu & Samways, 2003). Gibson, *et al.*, 1992 reported similar findings for spider diversity with an increase in diversity resulting from a decline in grazing intensity. At Stirling, all of these effects lead to simplified vegetation structure, and reduced grasshopper species richness and abundance. Three species, *Acrida accuminata*, *Calliptamulus natalensis* and *Calliptimicus semiroseus*, benefit from the over-utilisation of the grassland (Indicator Species Analysis) with increased abundance through changes in the vegetal structure and higher percentages of bare ground. Two species *Orthochtha dasycnemis* and *Qachasia fastigiata* showed a reciprocal trend. These species tend towards better veld (grassland matrix) condition areas, where the percentage of *Themeda triandra* was high and the grazing light or absent.

Structure plays a role in grasshopper assemblages and to a large extent is determined by utilisation. Structure however, appears to play less of a role in this instance than other authors have postulated (Joern, 1979; Gandar 1982b). In the Karoo as with this study the abundance of *O. dasycnemis* is affected by utilisation (grazing). It is absent from sites that are heavily grazed by livestock both in the Karoo and in KwaZulu-Natal Midlands, and is only common in areas where *Themeda triandra* dominates the grassland complex. In grasslands is devoid of large stands of *T. triandra* so are they devoid of *O. dasycnemis*, indicating their preference for *T. triandra*.

The rotational grazing regime at Goodhope 2 follows separate camps, with each camp being heavily utilised and rested for a long period before being re-utilised (Savory & Butterfield, 1999). Using this method of grazing enhances long-term benefits, with higher seedling recruitment in gaps (Bullock *et al.*, 1994). Grazing defoliation is distributed over a wider proportion of the plants, reducing selective grazing pressure. During the resting phase after defoliation, new tillers proliferate and fill the bare areas

created by patch grazing or excessive trampling. Therefore patches made bare during the first defoliation are occupied by new grass tillers, while dense stands of grass will be opened up (Onsager, 2000).

Such successional defoliation, by the movement of livestock from camp to camp, results in a heterogeneous structure within the grassland, comprising a range of plants with characteristic growth forms, structural attributes and chemical composition (Joern, 1979). Using rotational grazing also benefits the farmer with the weight gains and overall productivity of the livestock being improved (Onsager, 2000; Boyd *et al.*, 2001). Rotational grazing also prevents encroachment by undesirable plants and the removal of valuable perennial grass species.

A system of rotational grazing increases the number of different microhabitats that can be utilised by grasshoppers and can act in controlling pest species outbreaks (Samways & Lockwood, 1998; Lockwood *et al.*, 2000; Onsager, 1993, 2000). Several authors have shown that rotational grazing also improves the richness of other taxa, for example, grassland butterflies (Morris & Thomas, 1991, Swengel, 1996; Swengel, 2001), Coleoptera (Rushton *et al.*, 1990) and Hemiptera (Morris & Lakhani, 1979; Morris & Rispin, 1987). Grazing by domestic livestock has proved the best and safest way of controlling the encroachment of undesirable plants and maintaining biodiversity in South Africa (Shackleton, 2000). Rotational grazing is the most productive form of grassland utilisation benefiting both the invertebrate fauna and the farmer.

Cattle opportunistically grazed the third grassland site (Goodhope 1). The cattle are left to graze over an extensive area with no apparent control. At present the grazing pressure is low and does not appear to be detrimental, as the characteristic signs of trampling and 'patchiness' in terms of vegetation structure are not discernable. There were no patches where continual livestock aggregation took place. Grasslands in the KwaZulu-Natal Midlands are well vegetated by nutritionally high value grass species, unlike the Karoo where vegetation is patchy, sparse and adversely affected by the slightest grazing pressure. In the Karoo where livestock aggregate around watering points and shade trees, trampling and high-intensity grazing have a major impact on grasshopper assemblages (Gebeyehu & Samways, 2003), with this situation being mirrored in Zululand, where aggregation at water holes creates a shift in vegetation,

species assemblage and resultant changes in the soil dynamics (Rivers-Moore & Samways, 1996).

Shackleton (2000) found that vegetal species richness was higher on communally grazed grasslands than on protected area grasslands. Similar results were found in Spain by Verdu *et al.* (2000) who studied the effects of protection against grazing on the biodiversity within in national park. The species richness of the site may be increased through this method of grazing, but is greater species richness beneficial? The elevated species richness that occurs may introduce undesirable species that do not contribute to increased productivity in terms of livestock gains.

Grasslands at present are managed to increase the quantities of a certain suite of desirable grass species that will benefit the landowner. Muralirangan *et al.* (1993) pointed out that agricultural simplification of the landscape, by reducing the number of plant species, not only encourages the spread of one or more orthopteran species but also favours a more homogeneous spatial distribution of the species. Therefore by promoting the development of a certain suite of grass species through management suggests that grasshopper richness is likewise affected, as the results in this study have shown that increased grass species richness, habitat heterogeneity and plant architecture increase grasshopper species richness and abundance.

2. Wildlife grazing

Kruess and Tschamtkke (2002) found that sites that are grazed have lower species richness than ungrazed sites. In this study the contrary was found, with ungrazed sites and livestock-grazed sites having similar species richness, both being significantly higher than wildlife-grazed sites. At sites where no grazing took place the grasshopper densities were greatly increased (Capinera & Sechrist, 1982; Fielding & Brusven, 1995), which appeared to be the case in this situation, with the exception of Himeville.

Rivers-Moore and Samways (1996) found that livestock grazing was more detrimental than wildlife grazing, as the impact of livestock trampling is more localized than wildlife grazing. At Coleford and Himeville, the two species of wildlife are Black Wildebeest (*Connochaetes gnou*) and Blesbuck (*Damaliscus dorcas phillipsi*), which are

concentrate/selective grazers (Bothma *et al.*, 2002), choosing a particular suite of grass species to feed on. The habitat heterogeneity, in terms of vegetation height and structure, increases through this type of grazing. It could, therefore, be assumed grasshopper species richness would increase, as habitat heterogeneity has been shown to increase species richness. This however, does not appear to be the case as the unpalatable patches are not suitable for grasshoppers as the increase in vegetation height excludes light and heat, two essential facets required by grasshoppers for survival (Gandar, 1979).

At Himeville, trampling was highly localized around dung middens and the grassland was patchy in terms of vegetation structure, with areas of short grass interspersed with large tracts of long grass and a high proportion of bareground (10%). These communal middens are points of aggregation for wildlife, suggesting that they play a similar role to watering points and shaded areas for cattle in the studies conducted by Rivers-Moore and Samways (1996), Samways and Kreuzinger (2001) and Gebeyehu and Samways (2003) in more arid environments.

Grasshoppers tend to abandon areas of high disturbance i.e. communal dung middens as part of their ecological strategy (Kemp *et al.*, 1990), and move to more suitable habitats (Lockwood, 1993), where they are more assured of survival. If the high wildlife-stocking rates continue at Himeville, there may be severe reduction in species richness, with increased abundance of one particular species, *Crucinotacris cruciata* reducing this sites ecological integrity and role in future conservation planning. Cagnolo *et al.* (2002) showed that in the grasslands of montane Argentina, abundance, richness, diversity and biomass of insect assemblages were minimal in the most intensively cattle-grazed areas. In other studies by Samways and Sergeev, (1997), results showed a depression of populations of some species but also provided opportunities for outbreaks of others (Samways & Lockwood, 1998; Lockwood *et al.*, 2000; Onsager, 2000).

The butterfly assemblage was also affected by the level of grazing intensity with increased abundance and richness of generalists, particularly at Himeville. The most abundant species *Vanessa cardui*, a generalist of the highest order, utilises all forms of flowering plants as a nectar source. With the alteration of the habitat and the

dominance of *Vanessa cardui* came a reduction in species richness, and more importantly the disappearance of endemic butterfly species, that would traditionally have made use of the grasslands. The elevated abundances of *V. cardui* were associated with the encroachment of *Verbena bonariensis*, an alien invasive plant, onto the fringes of the grassland, where traditionally high levels of disturbance from wildlife movement and aggregation had taken place. A similar trend was noted amongst monkey beetles (Scarabaeidae, Hopliini) in the Karoo, which are pollinators influenced by levels of grazing, with an assemblage shift away from perennial and bulb pollinator guilds towards those favouring weedy annuals in overgrazed areas (Colville *et al.*, 2002).

Wildlife grazing mirrors the traditional type of land use, but due to human pressure, a reduction in acceptable grazing area has occurred, concentrating wildlife species onto smaller parcels of land, elevating grazing intensity, thus influencing grassland utilisation, and grasshopper and butterfly diversity. Wildlife grazing therefore mimics intensive continuous livestock grazing that has been shown in this and other studies to be highly deleterious to the grasshopper and butterfly assemblage and the grassland as a functioning ecosystem.

5.4.2. Disturbance: its effect on the grasshopper assemblage

The main contributing factor to disturbance within these grasslands is grazing by both livestock and wildlife and management practices. In South Africa, the most important contributor towards grasshopper abundance was disturbance from game or domestic livestock (Samways & Kreuzinger, 2001). The level or intensity of the disturbance ranged from slight to very intense. Linkages present in the forestry system may be firebreaks, access points or borders. Parts of these areas are often highly disturbed via over-utilisation of fire to protect the economically important crop, and the continual localized disturbance within the linkages by machinery that is required in running a forestry estate. At sites where levels of disturbance are highest, species richness is reduced. This reduction however is not necessarily mirrored by the abundance of species (Turner & Gardner, 1991). Some species apparently thrive on disturbed patches, as long as it is neither too severe nor frequent. This has been demonstrated in North America (Fielding & Brusven, 1993a; Porter & Redak, 1996 and Onsager

2000), in Russia, (Sergeev, 1998) and in South Africa (Gebeyehu & Samways, 2002). Most species that respond positively to disturbance are eurytopic (Sergeev, 1998).

At Himeville where grazing intensity and density of grazing species are both high, large areas are being denuded of the vital grass cover that is required by many species of grasshopper for survival. This alteration and reduction in microhabitats reduces the species richness (Onsager, 1993) with many species abandoning these areas (Lockwood, 1993). The reciprocal however is true for abundance. The alteration of the habitat favours certain species that are able to take advantage of this change, for example, *Crucinotacris cruciata* the dominant species at Himeville with 79% of total abundance.

1. Burning

Fire is an important component in the maintenance of grassland diversity (Freeman, 1998; Lunt & Morgan, 2002). In South Africa the most important form of defoliation is burning. Over the millennia natural fires have been responsible for shaping the structure and composition of grasslands in southern Africa (Frost, 1984; Hall, 1984; Scott, 1984, Tainton & Mentis, 1984; Armstrong *et al.*, 1996). The use of human-induced fire in management programmes mimics natural events to the benefit of the grassland biota. If utilised correctly, fire is able to control alien and undesirable plant species as well as rejuvenating the grassland. Fire removes dead organic material, allowing light and heat to penetrate into the soil, resulting in greater annual dry matter production (Ojima *et al.*, 1994; Blair, 1997; Snyman, 2002; Fynn *et al.*, 2003). Fire also reduces the number of parasites present within the grassland (Cully, 1999; Hardy, 1999). There are, however, drawbacks to the use of fire with the release of undesirable gases into the atmosphere (Fishman *et al.*, 1991) and posing a threat to economically important crops, especially forestry. Fire, if utilised too often, may cause a reduction in overall species richness and abundance across all taxa (York, 1999). Not allowing complete system recovery, results in the removal of vital leaf litter, moisture and simplification of habitat structure. The total exclusion of fire results in equally undesirable changes both in the grassland community and the organisms that it supports (Kruger, 1984; Tainton & Mentis, 1984).

Burning in this study did not have a negative impact on grasshopper assemblage richness and abundance, but did impact on the responses of individual species. Burning affects patterns of grasshopper population density and distribution both directly, through mortality, and dispersal and indirectly through induced changes in host plant communities (Gandar, 1982b; Evans, 1984, 1988a; Samways, 1990). Many authors have demonstrated that abundance of certain taxa, namely Orthoptera and Coleoptera (Carabidae) increase post fire (2-12 months) (Rice, 1932; Bulan & Barrett, 1971; Warren *et al.*, 1987; Anderson *et al.*, 1989; Bock & Bock, 1991; Greenslade, 1993; Reed, 1997). Anderson *et al.* (1989) found that there was no significant response of the Orthopteran assemblage post fire. Force (1981) showed a general decline in insect species richness, diversity and abundance during post-fire succession. The results here support this contention with a decline in species richness from biennially burnt sites through triennially burnt sites. The most species poor site was Wahroonga 2, which was never burnt. The decline in species richness may be explained by three possibly interrelated factors.

Firstly, burnt sites are likely to have less plant litter than sites that are burnt less frequently (Bigalke & Willan, 1984; Frost, 1984; Mentis & Tainton, 1984). The accumulation of litter reduces the availability and accessibility of suitable oviposition sites for grasshopper species (Mushinsky & Gibson, 1991).

Secondly, the grasshopper richness and abundance may reflect changes in live grass biomass, which is known to increase initially post fire (Evans, 1984, 1988a,b; Gandar, 1982b; Rowe-Rowe & Lowry, 1982). The continued absence of fire, promotes dead grass biomass and a reduction of live grass biomass (Robinson *et al.*, 1979). Grasshoppers benefit from regular burning, as the crude protein calcium and phosphate content are all higher in newly burnt grass in the growing season (Oliver *et al.*, 1978). Therefore the grasses at Wahroonga 1 (triennially burnt) and Wahroonga 2 (never burnt) are likely to be less nutritious and palatable than the regularly burnt grasslands.

Thirdly, with the absence of fire and or mowing, average grass height increased at the triennially burnt grassland. This was similar to the results found by (Chambers & Samways, 1998). As vegetation height increases and leaf litter accumulates, the

microclimate and particularly the temperature regime changes dramatically. The ground surface temperatures do not get as high as in short grasslands and the grass sward temperature is much slower to reach optimal temperatures. These two factors affect various facets of grasshopper biology, namely, egg development, metabolic rate, food consumption, digestibility and assimilation, adult life span and prey avoidance (Uvarov, 1977, van Wingerden *et al.*, 1991). These factors affect species that are unable to thermoregulate in long grass, as well as thermophilous species and cause them to be less abundant or even abandon the habitat.

Lamotte (1975) proposed that fire-induced habitat changes were important in determining insect composition. Post fire environments would favour species that prefer sunny, xeric and grass dominated systems. Species that show this tendency will be able to colonize these grasslands rapidly and take advantage of these conditions (Bock & Bock, 1991). Species that show different preferences requiring forb species to feed on would decrease (Evans, 1984, 1988a) and are eliminated from the system, as recently burned grasslands offer less niche diversity than previously unburnt grasslands as well as forbaceous plant species being highly reduced through burning (Porter & Redak, 1997).

In the United States, Porter and Redak (1996) found that species richness and abundance declined as a result of fire. The mitigation for the stark contrast to the results found in the United States was that in South Africa the grasslands were predominantly indigenous grass species that have adapted to fire. Porter and Redak (1996) showed that their grasslands were inundated with exotic grasses that did not respond well to fire, and the grasshoppers utilised these grasses as their major food source. After fire the exotic grasses were excluded and therefore so were many species of grasshopper. The numbers of forbaceous plants are also decreased through fire use. In the Porter and Redak (1997) study the dominant grasshopper species is a forbivorous species; hence it declined after fire, due mainly to the lack of edible vegetative matter. The opposite is true here with the dominant species being graminicolous, which benefit from fire, because the grass quality improves through the use of fire as it promotes regrowth and controls undesirable grass species. Chambers and Samways (1998) found that burning reduced the species richness and abundance when conducting their cafeteria experiments. These findings may be inappropriate in

assessing the responses of grasshoppers to fire, as the treatments were done in close proximity, which does not mirror the situation at the landscape level and at the level of this study.

The close proximity of these different treatments to one another allowed local migrations of grasshoppers to areas that were best suited to the needs of the grasshoppers at one particular time (Stebaev, 1970). Most insect species are vagile and opportunistic (Panzer *et al.*, 1995; Panzer & Schwartz, 1998; Panzer, 2002). At the landscape level the scale of these migrations is much larger, and for many species impossible. Many species sampled are flightless which prohibits large-scale migrations (Green, 1998), or are very slow to recolonize as they are weak-flying arthropods, and the fragmented nature of the landscape in which they live (Macdonald, 1989) prohibits free movement. In many instances plantations surround the grasslands acting as impenetrable barriers (Armstrong & van Hensbergen, 1996) to dispersal and migrations by both vertebrates and invertebrates. This was substantiated when a neighbouring *Pinus* sp. plantation at Goodhope 1 was sampled for grasshoppers, during the months of February and March which recorded six and 22 individuals respectively, comprising two species, compared with the neighbouring grasslands recording 188 and 466 individuals respectively, and representing 11 and 19 species respectively.

The size of the grassland and its distance from nearest neighbour are important, as this will influence the recolonization of the burnt grassland by grasshopper species. Recolonization is important as Gandar (1982a) showed that there is almost complete destruction of the arboreal insect fauna of trees in African savanna. This was mirrored at Karkloof 1 and 2 where there was total destruction of the grasshopper assemblage. This means that recruitment from other populations as well as hatching play an important role in the recolonization process. These results are well supported with what happened at the two Karkloof sites, which were completely devoid of species post-fire. Recolonization of Karkloof 1 was slow, firstly, due to the lack of close proximity source populations, and secondly, the time of year as the system was dormant. No nymphs were hatching and progression from the nymphal to adult life stage was taking place. In addition, to the highly fragmented nature of the landscape surrounding these two remnant grasslands was the size of the area burnt. The entire grassland patch was burnt at a high intensity. This can be deduced from the large fuel load (high number of

invasive forbaceous plants), the grassland is not utilised for grazing and was burnt in winter prior to any rain that may have reduced the temperature of the fire (Tainton, 1984c). If the grassland had only been partly burnt, recolonization would have been more rapid. At Linwood only half the grassland was burnt facilitating rapid recolonization, due to the proximity of a source population.

2. *Species responses to burning*

Fire seemed to increase the abundance and determine the presence of certain species, namely, *Eremidium basuto*. Where management excluded fire this species was absent from the grasshopper assemblage. Armstrong (Pers. Comm.) believes that this species' hatching may be triggered by fire. Panzer and Schwartz (1998) present observations of particular species or sites as corroboration that insects specialized to live in open habitats asserted to be fire-dependent would likewise be fire-adapted, which concurs with what happens for *E. basuto* and *Acorypha nigrovariegata tibialis* that are only present at sites which are recently and regularly burnt.

Three endemic species (*Dirshia abbreviata*, *Qachasia fastigiata* and *Whitea alticeps*) were recorded in their highest abundances at Wahroonga 2, indicating that even though fire is an integral part of grassland functioning (Acocks, 1988), certain species, in particular flightless endemics, tended to benefit in situations where the grassland was not burnt, as fire appeared to suppress their abundance. All three species are flightless and could not escape the effects of fire. By not burning the grassland these species were able to take advantage of favourable conditions and record elevated abundances. It can be concluded that to increase grasshopper abundances, particularly flightless endemics, the frequency of fire needs to be reduced or a different form of landscape management needs to be implemented. Swengel (1996) proposed that large tracts of grassland should not be burnt all at one time. In preference managers should burn small patches to form a mosaic effect within the grassland of burnt and unburnt areas. This will benefit both the fire dependent species, and the species that are restricted by their lack of mobility.

3. Season of burning

The time at which the grassland is burnt is important in determining the grass composition as well as the grasshopper assemblage. In the past grasslands were burnt during the winter period of June and July. Burning at this time has detrimental effects on the growth of grasses as they draw heavily on their reserves and when spring and the first rains arrive the vigour of the plant is highly reduced (Hardy, 1999). Burning during this period also reduces the amount of mulch (dead organic matter), which covers the ground and helps with rainwater infiltration into the soil. The survival of grasshoppers after burning depends to a large extent on the response of the vegetation. The regrowth of vegetation is dependent on the season of burning, the method of burning (head burn or back burn) and the suitability of the conditions (Tainton, 1984b; Tainton & Mentis, 1984; Trollope, 1984a). Research has shown that the ideal time to burn, for grass productivity, is early spring after the first 'good rains' (15mm in 24 hours) (Scott, 1955; Tainton, 1984b; Hardy, 1999). Burning at this time of year ensures that the soil surface is exposed to insolation for a minimum period, with runoff and erosion reduced to a minimum (Scott, 1955). 78% of the sites are burnt after the first spring rains, usually between the first and last week of September. Two sites (Karkloof 1 and 2) are burnt in June. The purpose being to prevent the spread of Curry's Post Weed *Phymaspermum acerosum* at Karkloof 1. June burning prevents seeds from germinating and reduces the recruitment rate of young plants and damages the mature individuals (Pers. Comm. D. MacFarlane). The control of *P. acerosum* is important, because if left it will dominate the grassland, resulting in a complete shift in the grasshopper species assemblage. The negative impact of winter burning results in grasshoppers being exposed to frost, with the reduced grass cover accentuating the thermal influences (Phillips, 1930; Samways, 1990) with similar findings for spiders (Riechert & Reeder, 1970). In this study Karkloof 1 and 2 recorded no individuals, which support the findings of Samways (1990), which showed that species with no refuge suffered 100% mortality. Four species (*Anthermus granosus*, *Faureia milanjica*, *Phaeocatantops sulphurius* and *Rhaphotittha cephalica*) are significantly affected by the time at which sites are burnt, showing significantly elevated abundances when burning is undertaken in July or left unburnt. This could be related to the time of hatching (*F. milanjica* and *R. cephalica*) and the ability of the species to escape the effects of fire (*Anthermus granosus* and *Phaeocatantops sulphurius*). Reciprocally,

Crucinotacris cruciata appears to be negatively affected by winter burning with its abundance reduced.

Spring burning on the other hand is advantageous to a certain suite of grasshoppers as many species are over-wintering (in the southern hemisphere) as eggs, resulting in many species being unaffected by fire. The survival of over-wintering adults and nymphs is dependent on the ability of the species to find enough food and shelter. Daubenmire (1968) showed that grasslands, which are burnt show superior regrowth when compared to grasslands that are unburnt. By burning at this time allow nymphs to hatch and not be decimated by fires later in the season as well as take advantage of the spring regrowth.

Winter and spring burns both have impacts on species abundance and richness. It is important therefore to take cognizance of this and develop a system where grasslands are not burnt at the same time each year. Burning consistently at the same time each year may favour certain species or communities at the expense of others (Duffey *et al.*, 1974; Mentis *et al.*, 1974; Kruger *et al.*, 1984; Tainton and Mentis, 1984; Gibson and Hulbert, 1987; Porter & Redak, 1996 and Chambers & Samways, 1998). Grasshopper species that are univoltine are more affected than bivoltine species (Swengel, 2001); as the time of burn may coincide with hatching therefore decimating the population. If this is undertaken on a continuous basis it may cause the species to be extirpated, as landscape fragmentation is taking place at such a large scale across the KwaZulu-Natal midlands, that source populations and the chance of recolonization is being reduced exponentially. Species with asynchronized breeding cycles or multivoltine species will be less affected (Swengel, 1996; Panzer, 2002). This was clearly illustrated here with certain species absent from grasslands burnt in July. In turn, certain species were only to be found in spring burnt grasslands. It is important for us to heed the conclusions drawn by (Daubenmire, 1968) that "in nearly every influence of burning, the time of year, or even the time of day, when the fire occurs is almost as important as the occurrence of fire itself".

4. Mowing

Mowing is not feasible throughout African savannas as only small proportions are conducive to mowing (Scott, 1970). In the KwaZulu-Natal Midlands mowing is not a very common form of grassland defoliation, due to the problems of access and in many cases extremely steep gradients that machinery are unable to operate on. Mowing has similar affects to burning in that it removes a large amount of dry matter from the grassland. Mowing is an advantageous practice, particularly in aiding species composition change of grassland. It has been well documented (Fynn *et al.*, 2005) that mowing promotes *Themeda triandra*, which is a desirable grass species as it has an exceptionally high crude protein percentage in comparison with other grass species naturally occurring in the KwaZulu-Natal Midlands (Bothma *et al.*, 2002). However, it does not have the benefits of burning as it does not control undesirable plant species, and often leaves a thick layer of dry vegetation that becomes matted reducing the number of bare ground patches utilised by grasshoppers for oviposition and basking. Chambers (1992) and Prendini *et al.*, (1996) found that annual mowing had a negative influence on the species richness of the grasshopper assemblage, which was substantiated by this study, however, their appeared to be no negative impact on the grasshopper abundance.

Fire tends to open up the grassland, by burning at different intensities forming a natural mosaic of grassed areas and patches of bare ground, which are utilised by grasshoppers as oviposition sites and basking areas. For the more motile species of grasshopper the type of defoliation is less important as they are able to migrate away from areas that are potentially hazardous and unfavourable. The reverse is true of flightless species that are unable to move rapidly and are often consumed by the fire before being able to breed and oviposit. In areas, which are mowed, forbaceous species like *Helichrysum aureonitens* are able to proliferate, due to their prostrate growth form, which allows them to avoid mechanical damage from mowing, unlike the more erect species with aerial meristems, which will be removed from the system. Avoiding mechanical damage allows them to grow, covering larger areas, providing greater habitat area for one species of grasshopper, *Qachasia fastigiata*. In addition, in order for this species to proliferate mowing may be essential to prevent the surrounding vegetation from growing up around it reducing its access to light, in turn affecting its

vigour (Fynn *et al.*, 2005). The season during which the grassland was mowed appeared to have an influence as *H. aureonitens*. This species flowers from July to February, avoiding any damage to its reproductive organs. At other sites it is regularly exposed to fire during flowering, reducing the amount of seed available for recruitment. Coupled with this is that at burnt sites, competition is greater due to the time periods between burns. The highest abundance of *Q. fastigiata* was recorded at the mowed site (Wahroonga 2), indicating that fire plays an indirect role in determining the grasshopper's presence as it appeared at other burnt sites. A combination of fire and competition from more erect grass and forb species suppresses the growth of *Helichrysum aureonitens*, which in turn appears to determine the abundance of *Q. fastigiata*.

Butterflies appeared to prefer fires to be excluded, as they are more prevalent at Wahroonga 2 and Wahroonga 1, which were mowed or burnt triennially respectively as opposed to biennial or annual burning, which was the management regime employed at the remainder of the sites. Burning impacts flowering plants abundance and presence/absence. Regular burning removes certain plant species, whilst promoting others. Many butterfly species have evolved 'hand in hand' with their larval host plant or food plant, loss of the host plant results in the loss of the butterfly species, which rely on that specific plant, thus reducing overall species richness.

Mowing does reduce the abundance of grasshopper species, but the recoveries of species abundance is rapid as many species move away from the immediate disturbance through sensing vibrations (Chambers, 1992) and later return to recolonize the area or make use of it. This was shown by the very small change in grasshopper densities, pre- and post mowing. Bulan & Barrett (1971) recorded similar findings with arthropod biomass recovering to pre-mowing levels within two weeks.

Through the creation of a homogeneous grass sward by regular mowing in late summer, has caused a decline in grasshopper species composition, by creating unsuitable habitat for late successional grasshoppers, namely, *Lentula minuta* and *Lentula obtusifrons* which require tall, complex environmental architecture or specific food plants that do not grow well under these modified conditions. Chambers (1992) showed that late season mowing (i.e. May as occurring in this study) exposes many

over wintering adults or nymphs to the more severe winter environmental conditions, thus having a negative impact on species abundance. In this study however, I would tentatively disagree with these findings as species richness declined appreciably during winter at all sites, with a very distinct and depauperate grasshopper assemblage being representative of the winter period.

Wahroonga 2 as with Coleford and Himeville sites showed high proportions of *Cruciotacris cruciata* and lower grass species diversity. Mowing simulates grazing and due to its uniform nature of defoliation leads to the simplification of the grass sward, with one species *Themeda triandra* being favoured (Fynn *et al.*, 2005). A late season mow can create a nutritious green flush that will favour *C. cruciata* in particular, improving the successful progression through the larval instars to adulthood.

Mowing played an important role in the species richness of butterflies and especially endemic butterflies. Wettstien and Schmid (1999) found that in Swiss montane wetlands, mowing supports a higher species richness than grazing, which agreed with this study, where mown sites were richer than a combination of fire and grazing. In grasslands that are frequently burnt reductions in larval abundances frequently occur, which is not mirrored in mown areas (McCabe, 1981; Swengel, 1995). In addition, to the direct mortality as a result of fire, many plants will not fruit or flower for the year preceding the fire, resulting in a reduction of biotope availability (Wright & Samways, 1998). The species that tend to be lost through the simplification of the butterfly assemblage as a result of fire are the grassland species and specialists. Haysom and Coulson (1998) showed a similar trend with uncommon moths increasing with an equivalent increase in time since last burning. Greater species diversity at mowed sites is as a result of greater larval recruitment with more larvae surviving the impacts of mowing than fire. Having said this however, the time of mowing needs to be carefully considered. Mowing during the middle of summer or in early summer reduces the number of feeding and breeding sites for invertebrates, when they would be actively searching out these vital resources (Kirby, 1992; Erhardt, 1995; Feber *et al.*, 1996; Baines *et al.*, 1998). Mowing at the end of summer (i.e. late May) is the most beneficial as most larvae will be in the process of over-wintering, reducing their movements and their need for host plants and nectar. In this instance the time at which this management tool is used will favour both the butterfly and grasshopper assemblage,

as disturbance is taking place at the end of the season, reducing potential mortality and exposure to detrimental conditions.

5.5. Sampling period, grasshopper richness and abundance

It is noticeable from the results that there is a definite shift in species assemblage through the period of one year. This change can be ascribed to a number of criteria, namely, temperature variation, food availability and competition. During the winter period, the number of species is highly reduced, indicating that these three criteria have a predictive ability in determining the grasshopper assemblage. In the summer period, the high grasshopper abundance in grasslands are predominantly due to two species *Conocephalus* sp. and *Orthochtha dasyncnemis*. These species are present throughout the warmer wetter part of the year, when the grass is greener and there is greater food availability. Many authors have postulated that grasshoppers are not restricted by their food preference to certain habitats, but rather by their resemblance (camouflage) to the vegetation within the habitat (Fogden & Fogden, 1974, Gandar, 1983; Gebeyehu & Samways, 2003). Both *Conocephalus* sp. and *O. dasyncnemis* are green in colour closely resembling the surrounding vegetation. Moreover, *O. dasyncnemis* appears to align itself along the long axis of grass blades, further camouflaging itself, providing greater protection against predators (Belovsky & Slade, 1993; Vitt *et al.*, 2000). Chambers (1992) showed that species such as *Lentula obtusifrons*, predominantly green in colour, occurs only in long grass. Species that are brown in colour tend towards areas that are a mixture of short grass and bare ground, which was mirrored in this study.

In the winter period, the dominant grasshopper species is *Cruciotacris cruciata*, which also shows similar behaviour to *O. dasyncnemis* in its alignment along the long axis of the grass blades. This species is well adapted to surviving in the drier, colder period of the year, showing very similar colouration to its surroundings. However, certain individuals that were captured during the wetter, greener part of the year were less brown and had obvious patches of green on their bodies. This suggests that grasshopper species are able to change their colour through responses to environmental factors, which are influencing them through their developmental stages (instars) (Ibrahim, 2001; Sword, 2002). The switch from the summer species to the

winter species demonstrates the different requirements of the grasshopper assemblage sampled. *C. cruciata* is found throughout the year, but its numbers are kept low in summer, presumably by competition from larger bodied species such as *O. dasyncnemis*, which through better camouflage (overall green colour) are potentially less exposed to predation pressure from birds and lizards (Belovsky & Slade, 1993).

5.6. Dominance of certain species

Two different suites of species, according to the sampling season, dominate the KwaZulu-Natal Midlands grasslands. Seventy percent of the recorded abundance is comprised of three species, two occurring exclusively in summer and one that occurs predominantly during winter. In the United States of America, grasslands appear to show a similar trait with one or two species dominating the grasshopper assemblage (Fielding & Brusven, 1993; Onsager, 1993; Porter & Redak, 1996), namely, *Melanoplus sanguinipes*. Gebeyehu and Samways (2002) showed a similar finding in the arid Karoo environment of South Africa with one species dominating the assemblage. In this study, the dominant species was *Crucinothraupis cruciata*, even though its numbers are highly reduced during summer. The high occurrence of *C. cruciata* (49% of total abundance) implies that it is the consummate generalist, with its only preference being for short and open grassland. It is a eurytopic species with little or no host-specificity, allowing it to take advantage of any conditions, namely conditions that are less desirable for other species of grasshopper.

5.7. Significance of elevated grasshopper abundance

Grasshoppers, being the predominant herbivores in grassland ecosystems, play a crucial role in ecosystem processes. Notably, they convert plants to nutrient rich frass. However, so do livestock and wildlife. The fundamental difference being in the size, which allows the more finely divided insect frass, to become available to plants more rapidly, than livestock or wildlife dung. Grasshopper frass is a vehicle for rapid recycling of nutrients (Belovsky, 2000), and with an elevation in their abundance, an escalation would occur in nutrient turnover. Boshoff (1988) postulated that grasshoppers will play an essential role in the long-term stability and optimal functioning of the Karoo. The Karoo differs greatly to the KwaZulu-Natal Midlands, but

the principal ecosystem engineers, grasshoppers, fulfill the same role of nutrient recycling in both systems.

As shown in this study, while grasshopper distribution and abundance patterns are principally governed by patch heterogeneity, amongst other factors, grasshoppers play a significant role in the regulation of ecosystem processes and as engineers creating landscape patterns.

Grasshoppers control the flow of critical resources and modify ecosystem structure (Belovsky, 2000; Gebeyehu & Samways 2003), and can therefore be considered 'webmasters' in both arid ecosystems and the grasslands of the KwaZulu-Natal Midlands.

CHAPTER 6

CONSERVATION BIOLOGY AND ECOLOGICAL ASSESSMENT

Conservation biology, being a young and vigorous science, is undergoing considerable methodological change. In an era of increasing conservation needs and tightening budgets, an increasing demand is that the scope of conservation management must be expanded to achieve economies of scale and efficiency (Simberloff, 1998). The first premise of landscape management is the maintenance of as much quality, natural, near pristine land as possible. This however is not possible with mans ever-increasing appetite for land, and his desire to manage and control the environment in which he lives. The goals of conservation management are proposed at 'maintaining biodiversity at current levels' (IUCN, 1991). This is biologically and practically unrealistic, as inevitably there are going to be more extinctions before a leveling out occurs. It is therefore of paramount importance that a method needs to be developed that will protect the natural areas that are left. At present the maintenance of every "cog and wheel"; the so-called art of intelligent tinkering (Leopold, 1953) is the most important facet of biodiversity conservation. The rivet-popper hypothesis, proposed by Anne and Paul Ehrlich, is in agreement with the art of intelligent tinkering, with all species playing a small and significant role within an ecosystem (Baskin, 1994). The pursuance of this thought process has been questioned by Walker (1992), stating that it is unwise to place equal significance on each species, as certain species are superfluous, and only a few key species are the 'drivers' maintaining the ecosystem. Many species are becoming extinct, both through direct impacts, and, more pervasively, through changes in ecosystem functioning, often because of multiple stressors, or synergistic effects (Samways, 2005). It is therefore imperative that answers need to be found to suppress such rapid loss of diversity and be able to maintain functioning ecosystems.

For insects, and much other biodiversity, the 'coarse filter' landscape approach is a critical approach for biodiversity conservation, where a manager normally has to consider a range of species, and their responses to different management and

environmental changes. This has been emphasized by Collinge *et al.* (2003) who showed that grassland type was the primary determinant of species richness of grassland butterflies in Colorado, USA, and that habitat quality was a secondary factor. In the case of prairie butterflies, living in indigenous remnants, 40% depend entirely on habitats unmodified by humans (Panzer *et al.*, 1995). The coarse-filter approach at present appears to be the most realistic approach, especially bearing in mind that less than 1% of the 10 million or so species have scientific names. The disadvantage of the 'coarse filter' approach is that it is blind to the way in which the immensely complex contents are being conserved (Haslett, 2001) and what each insect species, each evolutionarily significant unit and each insect polymorphism needs under all environmental conditions to survive. This is illustrated by the various movement patterns in a butterfly assemblage at any one point. Each species, although seemingly, simply flying by, is reacting sensitively to the various landscape vegetational structures (Wood & Samways, 1991). This serves to illustrate that conservation of insect diversity encompasses a vast complexity of interactions that in themselves vary over space and time. Against this background, the landscape may be considered as a continuously varying differential filter (Ingham & Samways, 1996), and try as we may, it cannot always be managed to provide optimal conditions for all species all of the time. This argues strongly for the conservation of larger spatial scales (i.e. landscapes and larger) such that all aspects of an insect's behaviour, and all types of insects can be facilitated. Nevertheless, conserving black boxes (i.e. whole landscapes with high connectivity, high ecological integrity and minimal human disturbance) is one answer in view of the magnitude of the biodiversity crisis and the shortage of time to conserve it.

By making use of the black box method of conservation enables the preservation of unknown species, this is particularly true of members of the invertebrate groups that have been marginalized due to their small size, even though they are the most highly adapted and rapidly lost species in the animal kingdom.

This however, is not to condemn the role of behaviour and the need for special, single-species, 'fine-filter' studies in special circumstances, especially when compiling the Red List (Hilton-Taylor, 2000). Single species management is dependent on knowledge of the habitat requirements of particular species, which may include some

disturbance, such as maintaining short and open grassland for the threatened lycaenid *Aloeides dentatis dentatis* in South Africa (Deuschländer & Bredenkamp, 1999) and maintaining large south facing slopes with dense stands of tall grass and the required host and feeding plants for *Orachrysops ariadne*.

A single definition of ecosystem management is not forthcoming with the word being differentially interpreted by the various parties that use it. Virtually all definitions of ecosystem management, focus on ecological processes rather than individual species (Meffe & Carroll, 1994). It is important to establish that the processes *per se* are not the valued entity, but the processes are believed to maintain the balance between the species and within the communities (Franklin, 1993).

6.1. Flagships, umbrella and keystone species

There is limited value attached to umbrella or flagship species, and they must be used with caution (Andelman & Fagan, 2000).

Flagship species such as the Richmond Birdwing Butterfly *Ornithoptera richmondia* (Horwitz *et al.*, 1999) and the Wart-biter Bush Cricket *Decticus verrucivorus* (Pearce-Kelly *et al.*, 1998) are usually physically large members of taxa that attract attention and garner sympathy. They are sometimes chosen on the basis of their dwindling population size or threat status (Dourojeanni, 1990). Their value for biodiversity conservation lies not so much in their ecological role but in their ability to perform strategic socio-economic roles (Walpole & Leader-Williams, 2002). For example, they attract public visitors to reserves, and in doing so, raise funds and local support for conservation thereby helping conserve wider biodiversity. Butterflies, such as Swallowtails (Collins & Morris, 1985) fulfill these roles among insects, with a reserve and site guide developed for viewing these taxa in Britain (Hill & Twist, 1998).

Many conservationists and managers hope that with the conservation of these species, a top-down effect will occur, in as much as they conserve other species and assemblages that overlap with their habitat requirements. However, this does not appear to be the case, with past conservation practices in South Africa having been blinded by public influence and attention, with most conserved areas having a flagship

species as their custodian. A classic example is the acquisition and protection of large tracts of savanna for the conservation of the White *Ceratotherium simum* and Black Rhinoceros *Diceros bicornis*. These large tracts of land do not incorporate diverse and heterogeneous units at the landscape level, rather creating a large homogenous area, with the various landscape attributes being repeated countless times across the landscape. Thus, many unique areas have been sacrificed due to the single-minded preservation of the two species of Rhinoceros, with poor management now threatening their future, as the encroaching alien invasive *Chromolaena odorata* (Triffid Weed) envelops more of the savanna each year, rendering these areas depauperate in biodiversity, non-productive and not fulfilling the goals of future conservation requirements.

Umbrella species themselves have been poorly defined, have been unproved in practice and may detract from wider ecosystem conservation priorities (Simberloff, 1998). An umbrella species is a 'protective umbrella' employed where the conservation goal is to protect a habitat or community of species in a particular area or type of habitat (Caro & Doherty, 1999). Usually it is a large species, often a mammal. Although such umbrella species have been widely advocated, there is virtually no proof of their value for over-arching biodiversity in general. Furthermore, the concern is that such an umbrella may be highly vulnerable itself, with its protective umbrella value being patchy and susceptible to dwindling. The term umbrella species seems therefore to have little currency; this is reflected in the protection of the Northern Spotted Owl (*Strix occidentalis*), whose habitat is artificially improved, to increase numbers of this species. By managing for this species, it is unable to indicate the 'health' of the system (Wilcove 1993; Tilman & Downing, 1994) it inhabits and thus its value as an umbrella species becomes redundant (Simberloff, 1987). However, the face of conservation is changing with a shift in thought processes to look for other avenues that will have greater returns in terms of biodiversity conservation than simple single-species conservation efforts. It is therefore imperative that conservation practitioners move beyond the flagship and umbrella species concepts, as they tend to make brash generalizations as to the richness of biodiversity within their spheres of influence. However they are fundamental in conservation as they provide a tangible example of conservation and continue to garner sympathy and crucial financial support for conservation from non-associated bodies and the population in general.

There are many systems, which are highly threatened, and receive no acclaim, as they have no poster child to represent them. Many systems with high potential conservation status, 'hotspots', have been lost through funds not being made available for their procurement, as well as too little research having been done to realize their potential importance as ecosystems. A South African example is Braamhoek. This unique escarpment wetland area situated on the border between KwaZulu-Natal and the Free State is extremely valuable to conservation efforts due to its size and pristine nature (Pers. Comm. D. Kotze). It is also a 'hotspot' for a large number of very rare and endemic birds (Pers. Comm. B. Taylor). This area has now been demarcated to receive a pump station for the generation of electricity. This will have dire consequences on the local avian community that utilises the wetland exclusively (Pers. Comm. B. Taylor). The effective loss of the wetland and surrounding grasslands will be compounded. With a loss of the birding 'hotspot' will come increased financial pressure on low-income communities that rely on 'twitchers' for their income. The resultant loss of income will create the need for the local communities to pursue other avenues of income generation and may lead to further degradation of the habitat that surrounds Braamhoek.

Many supposed umbrella/flagship species play no major role in ecosystem functioning and without them the ecosystem would still function, i.e. they are not keystone species (Simberloff, 1998). Keystone species is another term that is loosely applied. A keystone species could be considered as one whose impact on its community or ecosystem is disproportionately large due to its relative abundance (Power *et al.*, 1996). This concept however, has been criticized as it threatens to erode the utility of the keystone concept (Hunter, 2000b). Paine's (1969) original idea was that the species composition and physical appearance of an ecosystem are greatly modified by the activities of a single indigenous species high in the food web. Such a keystone species influences community structure and ecological integrity, with persistence through time. Mills *et al.* (1993) have pointed out that the term keystone has since been applied to a plethora of species, at different levels in food webs, and with very different effects, both qualitative and quantitative, in their communities. In terms of conservation, it is not so much the keystone species *per se* that is significant but its keystone role (Mills *et al.*, 1993). This returns us to the need for maintaining interactions as

advocated by many authors rather than the species as a single entity. Interaction strengths between species is a much more compelling concept than keystone species concept, and therefore more likely to be utilised. However, De Maynadier and Hunter (1994), argue that the term is beneficial as it helps to rally public understanding and protection for a system. If we are to utilise the keystone concept in practical conservation we need to understand the underlying implications in order for it to realize its potential.

6.2. Ecosystem management

The dilemma starts here. How do we manage ecosystems? Does one management tool encompass all facets of grassland management, the answer is no. Using one management system for the good of a specific taxa or species may prove detrimental to other taxa that are also corner stones of ecosystem functioning (Morris 1981a; Committee on Scientific Issues in the Endangered Species Act [CSIESA, 1995]). For example, the Ash Meadows (Las Vegas, U.S.A) naucorid *Ambrysus amargosus* is the only aquatic insect protected under the Endangered Species Act in the United States of America, and its population is still declining because of habitat alteration to favour the Devil's Hole pupfish *Cyprinodon diabolus* (Polhemus, 1993). In this study too, there is an example of one management practice-favouring the butterfly assemblage over the grasshopper assemblage. Wahrenoonga 2 recorded the highest butterfly diversity, but the poorest grasshopper diversity. Having said this, however, the management regime utilised favoured certain flightless endemic grasshopper species but not species that require fire to trigger hatching or provide ash, an essential element of their diet. Structurally the grassland became more homogeneous through mowing, reducing the variability of microhabitats required by certain later successional grasshopper species. Mowing over a long period and in consecutive years has simplified the grass species composition, further reducing microhabitat diversity. The elevated abundance of a number of forbaceous plant species present indicated that they benefited from fire exclusion. This would support the increased number of butterfly species found at the site by creating a greater food reservoir for exploitation. Therefore a management program and assessment needs to be developed that will look at a range of taxa within

the system and alter the strategies to best benefit the ecosystem as a whole and the landowner, who requires the land to be productive.

The problem with managing ecosystems is that they are highly site specific i.e. one grassland is not the same as another. Managers like to apply a single brushstroke to grassland management. The mindset of managers therefore needs to be altered, from a cellular view (e.g. Beef production) to a more holistic outlook (biological integrity integrated with productivity). By improving the ecosystem as a unit will have far-reaching consequences, both in terms of ecological value and productivity.

The principles and goals of management first need to be established before any success is achieved. There are three major facets to management. Firstly, the area and time frame of management need to reflect landscape-scale patterns and processes (Gutzwiller, 2002). Secondly, managing a landscape require goals that are fluid to accommodate natural spatial and temporal variability, and the uncertainty and surprise in outcomes that result from allowing ecological processes to take their course (Christensen, 1997). Thirdly, and probably the most important issue is that the management boundaries need too extend beyond administrative boundaries (i.e. all stakeholders need to 'buy' into the idea and manage areas co-operatively).

6.3. Rapid assessment

A rapid assessment technique will be able to provide the manager with the insight to alter practices over time, thus benefiting a wider range of taxa and improving the biological integrity of the ecosystem. The key facet of any management regime is that it must be fluid not rigid. It needs to develop symbiotically with the system that it is managing, responding to changes, especially in climate, the greatest threat facing conservation practitioners at present.

Most rapid assessment techniques that result in a model being formulated and utilised are in the aquatic context. Examples of these are available from a large variety of countries, namely the United States of America (Stein & Ambrose, 1998; Costanza & Mageau, 1999), Australia (Boulton, 1999), United Kingdom (Wright *et al.*, 1984) and South Africa (Chutter, 1994,1998; Dickens & Graham, 2002). These techniques have

been well received and widely used, in assessing the quality of the water within a stream or body of water. The main reason for these assessments being so enthusiastically received is that they are fast and cost-effective. They do not measure biodiversity at the species level, but rather relative abundance across taxa, which in turn indicate environmental stress on the system. Thus they monitor ecological health rather than finer aspects of ecological integrity (Resh & Jackson, 1993).

Why should a similar technique not be developed for use in grasslands, which will rate grasslands in a linear scale from highly modified to pristine? A rapid assessment tool will be able to predict how the system will change in response to management practices.

When developing an assessment technique there are a number of important determinants that need to be factored into the equation:

1. It must be cost-effective,
2. Is it an effective indicator of grassland quality,
3. Can non-specialists undertake it?

The grasslands of the KwaZulu-Natal Midlands are highly altered and continually threatened by anthropogenic advancements and perturbations. A multi-taxic approach is a more holistic approach to management, and will manage the ecosystem rather than a single focal group or individual species. Many people may argue that a multi-taxic approach will not be cost effective and will require intensive sampling and various sampling methods.

The taxa sampled were grass species, grasshoppers and butterflies. The first two species primary producers and primary consumers respectively, with the butterflies playing an important role in pollination in the KwaZulu-Natal Midlands (Field, 2002). Sampling of the grasshoppers can be undertaken at the same time as the butterflies while walking 50 metre transects. The grasses can be sampled by randomly choosing three hundred points within the grassland. This method has proved successful and representative of the entire grassland (Everson *et al.*, 1990).

A rapid assessment is a community characterization as opposed to a strict inventory (Longino & Colwell, 1997). Community characterization uses structured sampling to estimate the distribution of species abundance, community richness and complementarity with other communities (Colwell & Coddington, 1994). The community characterization aims at ranking habitats or parcels of land, or to assess them over time. In contrast, a strict inventory aims to obtain an accurate species list (Longino & Colwell, 1997). A strict inventory has merits when phylogenetic measures of conservation value are considered (Raven & Wilson, 1992). The aim of this study was to predict and monitor land management regimes as well as provide recommendations on, which avenues to pursue for managing biodiversity overall.

Species richness alone does not tell us about the dominance of one species and the rarity of another. This information is important as it provides a basis on how the assemblage as a whole is responding to a certain management practice. The use of species richness as a stand-alone index will not provide information on how the community structure is changing over time. The presence of a certain species does not substantiate the suitability of the management for a species, the presence of a large or dense population does imply this (Swengel, 2001), and therefore assessing the abundance of the individual indicator species is imperative. An example, from this study is *Crucinotacris cruciata*, which is common to all sites, but in varying degrees of abundance. The abundance of the species is determined by two variables, the time at which the grassland is burnt and the intensity of the grazing pressure. A presence-absence study alone would not be able to furnish the manager with this information. Grasslands in the KwaZulu-Natal Midlands are increasingly being fragmented, with many areas only retaining the common species, as the more specialist interior species are being lost because the size of the grassland fragments are unable to sustain them. This therefore shifts the onus of importance from species richness to species abundance in order to monitor the deleterious effects of anthropogenic influences, through habitat fragmentation or poor management.

All grasslands in the KwaZulu-Natal Midlands are subjected to some form of management and monitoring is required to assess the management regime imparted. The most appropriate method of tackling the problem of monitoring is to use community characterization. The objective of any assessment technique is to find

particular repeatable measures that accurately reflect the patterns in the underlying ecological community. This approach has been exemplified by the work of Oliver and Beattie (1996) who sought out ways to make assessment more efficient using surrogate sets and recognizable taxonomic units (RTU's/morphospecies).

There has been much debate, especially related to aquatic systems, as to what taxonomic level (either family or species level) is most suited for biological monitoring (e.g. Guérol 2000; Lenat & Resh, 2001). The consensus is that, wherever possible the sorting to species is better (Austin, 1999; Panzer, 2002; Bassett *et al.*, 2004). However this may not always be possible (Bailey *et al.*, 2001) and when working at broader geographic scales, using higher taxa appears to be better suited to the task.

In this study the sorting to species level was possible, as species diversity among the three focal taxa was low in comparison to similar studies undertaken in more tropical and hence more speciose areas. Bassett *et al* (2004) have shown that studies based on terrestrial arthropods, sampled as orders or guilds, distributed along disturbance gradients and confined to small geographic areas only achieved a poor discrimination of sites. With the increased resolution of the hierarchy (i.e. the change from using guilds to families) improved the indicator ability of the sampled invertebrates. In the case of this study, species sampling, and the use of named species allowed for the collection of absolute abundance, thus fine-tuning the focal taxa's ability to provide a more holistic picture as to the dynamics and responses of the sampled assemblages within the grasslands studied.

In this study a surrogate sample set of grasshoppers has been teased out of the full grasshopper sample set. These surrogate grasshopper species represent a range of families, feeding guilds and respond either positively or negatively to vegetation and management regimes. The grasshopper surrogate set chosen here answers the question posed by Jones and Eggleton (2000) "How well does a sample of the target taxon characterize the variation within its own group?"

6.3.1. Surrogate choice

22 grasshopper species were chosen to fulfill the role of surrogate species for the entire grasshopper assemblage. These species showed a significant correlation when compared with the full sample set (Mantel's $r = 0.90$ and $p = 0.001$). In terms of grasshopper abundance the Mantel's test showed a stronger correlation between the surrogate set of species and the full compliment that were recorded (Mantel's $r = 0.99$ and $p = 0.001$). These species represent a saving in time and cost of the assessment technique. The surrogate species were chosen using six criteria:

1. The abundance of the species at each grassland patch,
2. The spatial distribution of the species across grassland patches,
3. Endemicity (six of the 22 species),
4. Specificity to certain habitat types,
5. Feeding guild representation
6. Response to management practices and disturbance.

1. Families

The species chosen were easily identifiable, common and belong to six families of grasshopper. The Acrididae comprise 71% of the total species richness of these grasslands. Fourteen of the 22 species (64%) belong to the Acrididae in the surrogate species measure. Lentulidae were represented by four species, with the remaining four families were represented by one species each.

2. Functional groups

The 22 species were representative of the different feeding guilds associated with KwaZulu-Natal Midlands grasslands, representing all four feeding guilds, namely:

1. Forbivorous species,
2. Graminicolous species,
3. Grass eaters and
4. Ambivorous species.

The representation of all the feeding guilds would prevent the surrogate set from being one dimensional, in that it only represents grasshopper species that utilise grasses and discounts the Forbivorous and Ambivorous species. These species were important as many South African grasslands have substantial forbaceous plant contributions. Forbaceous plants are an integral component of grasslands, adding to the structural heterogeneity of the grasslands and there overall biodiversity (Baskin, 1994).

6.3.2. Choice of grasshopper species as indicators

A total of 33 species of grasshopper were statistically significant, and, indicative of various environmental and management regimes at $p < 0.05$. Of these 33 species, only 22 species were utilised in the surrogates' matrix. The 22 species were the most representative, abundant and environmentally responsive species of the entire grasshopper assemblage. The reasons for the exclusion of 11 species are discussed below.

Eight species of grasshopper were excluded as they were only encountered at 30% or less of the sites sampled. The reasons for the exclusion of these species were that (a) they may have been poorly sampled by the collection technique; or (b) they may have been collected from marginal habitats (Novotny & Basset, 2000). When developing an assessment tool, cognizance must be taken of two important variables that relate to the grasshopper species. Species presence/absence across sites and the species abundance at sites determine whether the species would fulfill the role of indicator species, as assessment is about utilising repeatable measures.

In the case of *Acorypha nigrovariegata tibialis* it occurs at three of the ten sites. It had a moderate abundance of 88 individuals. Having said this, its distribution across the sites was highly skewed, with 76% of individuals occurring at one of the three sites. It was excluded on the grounds that when sampling for the rapid assessment model, it would be unlikely that this species would be regularly encountered, therefore skewing the results of the assessment model in the favour of sites where it was present.

Six species, *Aneuryphymus montanus*, *Calliptamulus natalensis*, *Cannula gracilis*, Gryllidae, *Gastrimargus determinatus vitripennis* and *Ornithacris cyanae* were omitted

because they were encountered at three or less sites and recorded abundance values of less than 20 individuals across the sites at which they were encountered. *Heteropternis guttifera* was excluded on the grounds that it was only found at 40% of the sites and was recorded six times.

Oxya hyla hyla was excluded as it was only encountered at one site Goodhope 1. It was abundant at this site, having recorded 134 individuals. It was indicative of the wetland/grassland interface at this site, as it was only recorded at three transects that traversed the edge of the wetland area. This was its preferred habitat with its modified tibiae for swimming.

Two species *Gastrimargus drakensbergensis* and *Orthochtha rosacea* were excluded, as they were indicative of the same variables as other more abundant species, namely, *Cruciotacris cruciata* and *Orthochtha dasycnemis* respectively.

Grasshopper species either responded positively or negatively to the variables tested. The grasshopper responses were separated into three distinct sub-groups, which represented the grasshopper species response to topography, vegetative characteristics and management regimes.

1. Topography

In the case of topography, only one variable was tested with eight species being indicative of slope aspect. Seven species were significant in terms of their orientation, preferring the cooler southern slopes as opposed to the warmer northern, eastern and western slopes. Three species, namely, *Acorypha nigrovariegata tibialis*, *Oxya hyla hyla* and *Aneuryphymus montanus* were omitted due to low abundance and/or poor distribution across sites. Two species *Lentula minuta* and *L. obtusifrons* showed a strong association with poor veld condition as well as high forbaceous content at sites, which tended to group together during Canonical Correspondence Analysis (CCA), as high forbaceous content usually represented poor veld condition. It must be stated that these two species were forbivorous species. South facing slopes in the South African context, tend towards a higher forbaceous content, and if left unmanaged would, over time, become natural forest fragments (Acocks, 1988). Bearing this in mind it could be

deduced that these two species are equally indicative of forb rich grasslands and a southerly aspect.

2. Vegetative characteristics

Nine of the 21 species that were indicative of vegetative characteristics were only indicative of one of the four categories. Five species (*Acorypha nigrovariegata tibialis*, *Gastrimargus drakensbergensis*, *Heteropternis guttifera*, *Orthochtha rosacea* and *Oxya hyla hyla*) were omitted for the reasons mentioned above. Seven of the remaining 16 species were indicative of the division between Highland Sourveld and Midlands Mistbelt grasslands.

Two species were indicative of poor veld condition, *Cyrtacanthacris aeruginosa* and *Lentula minuta*. Both of these species were forbivorous species and preferred sites dominated by forbaceous plants. In general the higher the forb content the lower the key and benchmark veld condition in these grassland patches. The validity of *Lentula minuta* as an indicator of poor veld condition is questionable, as it merely indicates a higher forb content, which only impacts on the grassland productivity in terms of livestock utilisation. *Cyrtacanthacris aeruginosa*, on the other hand is predominantly found in rank vegetation with long grass or in disturbed areas such as agriculture and gardens (Picker, Griffiths & Weaving 2002), which would indicate a change in veld condition for the worse, and validate its ability to indicate poor veld condition. This could be stretched to encompass areas, disturbed by anthropogenic influences.

Ochrophlebia caffra was indicative of grasslands characterized by low forbaceous content, short grass sward and good veld condition. In addition, this species was also indicative of Highland Sourveld grasslands. *Calliptimicus semiroseus* was also indicative of low forb content grasslands, which were in good condition, in terms of the key grass species assessment. *Eremidium basuto* indicated grasslands with low forb content, as well as preferring Highland Sourveld grasslands to Midlands Mistbelt grasslands. This would be expected as this species is classified as a montane endemic (Pers. Comm. Armstrong, 2004; Foord *et al.*, 2002). *Eyprepocnemis plorans* showed the reverse of *E. basuto*, indicating high forb content and a preference for Midlands Mistbelt grasslands.

Table 37. The 22 species of grasshopper utilised to assess the various management regimes and responses to environmental disturbance. (The abundance of each of the species will need to be recorded in order to assess the intensity of the impact or perturbation)

Grasshoppers
ACRIDIDAE
<i>Acrida acuminata</i>
<i>Anthermus granosus</i>
<i>Calliptimicus semiroseus</i>
<i>Catantops melanostictus</i>
<i>Cruciotacris cruciata</i>
<i>Cyrtacanthacris aeruginosa</i>
<i>Dirshia abbreviata</i>
<i>Eyprepocnemis plorans</i>
<i>Faureia milanjica</i>
<i>Gastrimargus crassicollis</i>
<i>Gymnbothrus temporalis temporalis</i>
<i>Orthochtha dasycnemis</i>
<i>Phaeocatantops sulphurius</i>
<i>Rhaphotittha cephalica</i>
LENTULIDAE
<i>Eremidium basuto</i>
<i>Lentula minuta</i>
<i>Lentula obtusifrons</i>
<i>Qachasia fastigiata</i>
PYGOMORPHIDAE
<i>Ochrophlebia caffra</i>
TETTIGONIIDAE
<i>Conocephalus</i> sp.
TETRIGIDAE
<i>Phloeonotus humilis</i>
THERICLEIDAE
<i>Whitea alticeps</i>

Cruciotacris cruciata stands out as an exceptional indicator species, indicating grasslands that have low forbaceous plant material (LOW 54, HIGH 19 $p = 0.01$), short grass sward (SHORT 70, TALL 5 $p = 0.00$) and good veld condition (GOOD 61, POOR 12 $p = 0.00$). *C. cruciata* showed no significant preference for either Highland Sourveld or Midlands Mistbelt grassland, reinforcing its wide distribution and its more generalist behaviour. *Orthochtha dasycnemis* is an exceptional indicator of short grass sward (SHORT 60, TALL 5 $p = 0.00$) and grassland type (MIDLANDS MISTBELT 52, HIGHLAND SOURVELD 9 $p = 0.01$).

3. Indicators of management

Two species were indicative of three management categories. The species were *Acrida acuminata* and *Qachasia fastigiata*. *A. acuminata* was indicative of grasslands that were grazed, highly disturbed grasslands (Himeville and Stirling) and preferred grasslands that were burnt during the springtime. *Q. fastigiata*, a flightless endemic, preferred grasslands that were not disturbed by grazing and not burnt or burnt triennially. In this case, it could be assumed that this species preferred grasslands that were not burnt as the highest abundance of this species was encountered at Wahroonga 2.

Grazing

Only four species of grasshopper responded to grazing categories with two species preferring grasslands that were not grazed, *Catantops melanostictus* and *Conocephalus* sp., with the other two species (*Gymnobothrus temporalis temporalis* and *Ochrophlebia caffra*) preferring grazed grasslands. *C. melanostictus* and *Conocephalus* sp. preferred grasslands where the sward was dense and moisture was retained during the heat of the day. *G. temporalis temporalis* preferred open areas with patches of bare ground, as well as road verges, and grasslands that were burnt more regularly, opening up the grass sward and creating more bare ground areas.

Season in which the grassland was burnt

Seven species were indicative of when grasslands were burnt. Six species preferred the grasslands to be burnt during winter. From this we could tentatively conclude that there was high nymph mortality from spring burning, which coincides with hatching, leading to poor recruitment rate. Nymphs may not die as a direct result of the fire, but may die from food shortage post fire, as the available resources have been significantly reduced as well as exposure (Samways, 1990) and increased predation pressure (Belovsky & Slade, 1993). However, in the case of *Cyrtacanthacris aeruginosa* it was difficult to assume that this species was responding to the time at which the grasslands were being burnt, as this species was most common at Karkloof 1. This species was prevalent at Karkloof 1 not in response to the burning regime but rather due to the nature of the vegetation at this site (i.e. high forb content and rank grassland).

One species *Acrida acuminata* preferred the grasslands to be burnt in spring, as this appeared to have no influence on its nymphs as the adults tended to be recorded later in the season (January to May) demonstrating that hatching in this particular species tended to occur only once regrowth was sufficient to preclude elevated levels of predation and lack of food resources for growth and development of the nymphs.

Biennial burning versus other forms of defoliation

One species was indicative of biennial burning, *Phloeonotus humilis*. This species was totally absent from sites that were not burnt i.e. mowed (Wahroonga 2) or burnt triennially Wahroonga 1. This species appeared to prefer grasslands that had a patchy distribution of vegetation interspersed with areas, which were bare. Fire produces this mosaic affect, with certain patches more prone to the vagaries of fire. This gave rise to more structurally heterogeneous grassland. Mowed grassland, was very homogeneous, due to the even nature of defoliation. The same can be said for the triennially burnt grasslands. The triennially burnt grassland has a much greater fuel load than grasslands burnt biennially. This fuel load allows for much hotter fires, which will burn more uniformly throughout the grassland, thus causing it to be more homogeneous in terms of plant structure.

Four species, *Dirshia abbreviata*, *Eyrepocnemis plorans*, *Qachasia fastigiata* and *Whitea alticeps* were indicative of alternative regimes i.e. mowing and triennial burning. Three of these species were flightless endemics and as mentioned earlier were more prone to the effects of fire, as they were not able to escape the fire. The fourth species *E. plorans* was most common at Wahroonga 1, a site with tall grass and a relatively high forb content.

Disturbance

The first site, Himeville, was over-utilised by Blesbuck (*Damaliscus dorcas phillipsi*) and Black Wildebeest (*Connochaetes gnou*). The area, which these species were able to utilise, was small c.a. 4Ha. The number of animal units was 25 animals, or 6.25 AU/Ha. (Pers. Comm. A. Armstrong) reinforced this observation, agreeing that this area was over-utilised. The second site was Stirling, which was highly disturbed

through excessive grazing by cattle. This area was traversed by many cattle tracks, with the cattle appearing to freely graze the area. This can only be assumed, but out of 12 visits to the site cattle were utilising this grassland on 11 occasions. Two species responded to this high disturbance level, namely, *Acrida accuminata* and *Calliptimicus semiroseus*. *Orthochtha dasyncnemis* and *Qachasia fastigiata* preferred sites that were not disturbed. Only 4% of individuals of *O. dasyncnemis* were recorded at the two disturbed sites. *O. dasyncnemis*'s absence or very low abundance points to this species being a good indicator of disturbance (LOW 54, HIGH 4 $p = 0.02$). The study undertaken by Gebeyehu and Samways (2003) reinforces this as they demonstrated the same presence and absence across a gradient of disturbance.

The above mentioned species were indicative of various facets of grassland structure, slope orientation and management regimes and will fulfill the role of surrogates for the grasshopper assemblage, forming the basis for a rapid assessment technique. The grasshoppers proved to be very useful as indicators of management practices and environmental variables. Bearing this in mind, it could be postulated that grasshopper species were the ideal taxon to study as they were abundant, showed significant species richness differences across grassland sites and were easy to capture, identify and sampling time would be highly reduced, which makes the sampling of grasshoppers cost-effective as well as representative of the quality of grasslands. Having said this, however, it would not be pertinent to utilise them as a stand-alone measure, but in tandem with the grass species assemblage. Grasses and grasshoppers together provide a metric for rapid assessment of the quality of grasslands in the KwaZulu-Natal Midlands. While it is not claimed here that this finding is definitive, it does provide a useful starting point for assessing grassland condition and for making recommendations for grassland improvement. During such improvement it is essential to monitor the grass and grasshopper assemblages and to assess changes in assemblage composition. When conditions improve, not only will there be a tendency towards increased species richness and greater proportion of rarer specialist species but also an increase in general grasshopper abundance. Table 38 summarizes this metric.

Table 38. Grasses and grasshopper species characteristic of grasslands that fall into one of three categories, i.e. poor, moderate or excellent. The grasses are sampled using a random three hundred point sampling technique. The grasshoppers are sampled by walking ten fifty metre transects through the grassland. The period in which sampling is undertaken is important because of the species richness and distribution across the species. The best time to sample grasshoppers is in February and March (greatest species richness), with the best sampling time for grasses being February, when inflorescences are present. It is important to note that this is not a definitive guide to grassland condition, but is based on the findings of this study and directs us towards certain conclusions

Grassland Status	Characteristic grass species	Characteristic grasshopper species	Comments on abundance of grasses and grasshoppers
Poor Condition			
	<i>Alloteropsis semialata</i> (a)	<i>Acrida acuminata</i> (1)	All grass species increase in abundance as grassland condition deteriorates, with (b, d, f, g) dominating the grassland Species 4 traditionally increases in abundance as condition deteriorates in short grass sward grasslands. Species 1, 3 and 5 usually increase in response to higher forb content and a taller grass sward in debilitated areas
	<i>Aristida junciformis</i> (b)	<i>Calliptimicus semiroseus</i> (2)	
	<i>Cymbopogon excavatus</i> (c)	<i>Catantops melanostictus</i> (3)	
	<i>Eragrostis curvula</i> (d)	<i>Gymnobothrus temporalis temporalis</i> (4)	
	<i>Miscanthus capensis</i> (e)	<i>Ochrophlebia caffra</i> (5)	
	<i>Sporobolus africanus</i> (f)		
	<i>Sporobolus pyramidalis</i> (g)		
Moderate Condition			
	<i>Heteropogon contortus</i> (h)	<i>Conocephalus</i> sp. (6)	The three grass species are more common and in certain instances, (i) will be the dominant species. The two grasshopper species are low in abundance with less than 30 individuals per ten transects.
	<i>Hyparrhenia hirta</i> (i)	<i>Cruciotacris cruciata</i> (7)	
	<i>Eragrostis racemosa</i> (j)		
Excellent Condition			
	<i>Bracharia serrata</i> (k)	<i>Conocephalus</i> sp. (6)	The grass sward is traditionally dominated by (o) with relatively equal distributions of the remaining five species. The grasshopper assemblage demonstrates elevated abundances of all five species with 6 and 8 recording over 300 individuals per set of ten transects, during February and March. Abundance of species 10, 11 increases in response to less grazing and reduced fire frequency
	<i>Diheteropogon amplexans</i> (l)	<i>Cruciotacris cruciata</i> (7)	
	<i>Monocymbium cerissiforme</i> (m)	<i>Orthochtha dasyncnemis</i> (8)	
	<i>Setaria sphacelata</i> (n)	<i>Gastrimargus crassicollis</i> (9)	
	<i>Themeda triandra</i> (o)	<i>Qachasia fastigiata</i> (10)	
	<i>Tristachya leucothrix</i> (p)	<i>Dirshia abbreviata</i> (11)	

6.3.3. Choice of butterfly species as indicators

The only significant compositional difference found was when comparing the season when grasslands were burnt. This is important as it demonstrates the vulnerability of butterfly species to burning at particular times during the year. Five butterfly species were indicative of when the grasslands were burnt. Three species preferred the grasslands to be burnt during the winter period, and one species in spring. All these species were most abundant at Karkloof 2, a winter burned site. Three of these species, *Aloeides oreas*, *Orachrysops lacrimosa* and *Pseudonympha varii* utilised grassland patches exclusively with the other two species, *Belenois aurota* and *Papilio demodocus* appeared to fly over the grasslands, and stop infrequently to utilise plant species within the grassland patch.

Butterflies have been shown to be good indicators of biological integrity in many studies, predominantly European studies, (examples Marshall & Haes, 1988; Thomas, 1991; Zchokke *et al.*, 2000) and an American study (Swengel, 1996).

At present this taxa does not provide a true reflection of biological integrity, due to the low abundances, low species richness and the nature of the species encountered in this study. Similar results were obtained from the Fynbos with butterfly abundance too low to provide insight into the processes that were impacting upon the Fynbos (Pers. Comm. M.J. Samways). In many instances only a certain stage in the butterfly lifecycle would make use of the grassland. Most species were in fact not grassland specialists, but generalists, utilising a range of habitats, from roadsides to the grassland/forest interfaces.

CHAPTER 7

CONCLUSIONS & MANAGEMENT RECOMMENDATIONS

7.1. Conclusions

7.1.1. Responses to structural heterogeneity

The type of grass sward and the general characteristics exhibited by the grassland play an important role in determining the grasshopper species assemblage. Grasshopper species preferred more open, short grass swards, where they were able to access areas of light and shade easily, thus regulating their body temperature. Tall dense grass swards tend to prevent the formation of open bare areas that can be utilised as basking sites, areas of aggregation and oviposition sites. Species of the Acrididae were noticeably absent from these tall dense grasslands, with the converse true for members of the Lentulidae, which preferred the taller, damper and more shaded grasslands.

7.1.2. Responses to grazing

Livestock and wildlife grazing both have a deleterious affect on grasshopper species richness, if intensively utilised.

Intensity levels need monitoring, as over-utilisation has resulted in a reduction in species diversity, through excessive trampling and a resultant shift and simplification (Van Wieren, 1998) in grass and plant species composition. It is therefore important that correct stocking rates (SR) are adhered too, and at some sites, namely Stirling and Himeville, the stocking rates need to be revised and reduced.

The best form of grazing for commercial returns is a rotational grazing system, which regularly rests camps and creates heterogeneous landscape utilisation, thus benefiting the grasshopper assemblage, as well as other invertebrate taxa (Swengel, 2001). The wildlife grazed sites averaged the lowest species richness.

It is imperative wildlife grazing practices in the KwaZulu-Natal Midlands be reassessed in order to reduce their impact on grasshopper assemblages, as most grassland

conservation areas are exposed to continuous large ungulate grazing, endangering their role as conservation areas and threatening their status as hotspots and havens for biodiversity. Conservation authorities need to shift their focus from large visible ungulate conservation to the smaller, often more important drivers of grassland ecosystems.

7.1.3. Responses to fire

The most species diverse grasshopper assemblages were recorded at grasslands exposed to some form of fire management. Where burning was utilised less frequently (triennially), the species richness and abundance was markedly reduced, in response to the development of unfavourable environmental conditions. At sites where fire was completely excluded the lowest grasshopper diversity was recorded, but conversely the highest butterfly diversity. Fire exclusion appeared to have a noticeable effect on the abundance of individual species (Bei-Bienko, 1970; Chambers, 1992) with certain brachypterous species showing elevated abundances.

The period in which the grasslands are exposed to fire has an impact on the species assemblage, abundance and ability to recover. The burning of areas over many seasons at the same time of year has had an influence in molding the species assemblage. Burning after the first spring rains appears to be detrimental as many species' hatching response is triggered by rainfall, and therefore the most vulnerable period of their life history is exposed to fire and the synergistic effects as a result of fire.

Regular burning, i.e. biennial burning is appearing to be impacting on the butterfly assemblage, with a reduction in specialists and an increase in more generalist species (e.g. *Vanessa cardui*). Where fire is excluded for longer periods butterfly richness is markedly elevated.

7.1.4. Responses to mowing

The mowing of grasslands creates a scenario where there is no successional advancement, and so places restrictions on the suite of grasshoppers that will be encountered. Mowing also tends to influence the richness of grass species that will

proliferate simplifying and reducing structural heterogeneity (Fynn *et al.*, 2005). The more eurytopic and stenotopic species will favour conditions produced by this type of management. Mown areas will therefore exclude those species, which require late successional plant species or architectural diversity within the plant community.

An increased diversity in butterflies indicates that they are able to take advantage of the situation, with the abundance of certain nectar plants proliferating, thus creating an attraction value. Coupled to this is that areas which are exposed to frequent fires might lose many plant species, resulting in a depauperate plant assemblage and thus a narrowing and simplifying of the grassland patch (Wright & Samways, 1998, 1999). I would hypothesize that it is not the mowing *per se* that influences butterfly species richness, but a combination of fire exclusion and the period during which the mowing takes place (late summer).

7.1.5. Assessment of grasslands

A subset of grasshoppers teased out from the species assemblage will be able to identify management practices and indicate future management paths for the grasslands. This subset of species will allow for an assessment to be undertaken that is simple, rapid and cost-effective (small number of species, easily identifiable and only 21 fifty metre transects need to be sampled in order to achieve a 95% C.I. for the surrogate grasshopper assemblage), and in conjunction with a random three hundred-point grass survey will provide the practitioner with a management tool that can be used as a guideline to alter management in response to the various regimes that are imparted on the receiving environment.

The utilisation of this type of assessment will also provide conservation practitioners with a tool that will rapidly assess the health of the grassland, and determine whether or not it will be able to be rehabilitated or play a role in future conservation goals.

7.2. Management recommendations

7.2.1. Has past management affected the species assemblage?

It is of paramount importance that management is ongoing as a site changes continuously, usually as a result of succession (Morris, 1991). This is true of the South African situation, where grasslands are not considered climax vegetation types (Acock's, 1988). However, by managing the system, an impact that is more than simply maintaining the ecological status quo will be imparted. Inevitably management plays a role in 'contemporary evolution' (Stockwell *et al.*, 2003). In this instance management can be viewed as a type of habitat destruction, with old, little disturbed, mid and late-successional habitats declining as a result of management preventing succession, which is traditionally unfavourable for the rearing of livestock. On the other hand, the amount of disturbed early-successional habitats increases. Presently we are managing an already altered landscape mosaic that is reflecting our historical management choices? This potential for management-influenced evolution is emphasized by the effect that burning management has on ants in Australia (Vanderwoude *et al.*, 1997). Therefore, a balance needs to be maintained. In this study, it appears that frequent, same season burning and livestock grazing have impacted upon butterfly richness and abundance. The low richness and abundance has created a situation that excludes butterflies from fulfilling the role of indicators of environmental change in this situation. The grasshopper assemblage appears to have been influenced by frequent fire, with many of the fire susceptible species excluded from site-specific grasshopper assemblages.

7.2.2. Grassland management to improve biodiversity

The environment is ever changing in response to global warming as well as relentless human influence (Bourn & Thomas, 2002), and one must realize that not all species will survive. One must assume that past management has had an influence on the species assemblage, with the current assemblage being a subset of the original. The current subset may be more depauperate than the original or simply compositionally altered from the original assemblage. Therefore species required to maintain ecosystem functions, as they currently exist need identification and management to prevent the collapse of vital ecosystems, grasslands being one of them.

Most of the extensive grasslands encountered within the KwaZulu-Natal Midlands fall within the realm of private ownership and are traditionally used for animal production.

Therefore, it is best to assess the methods that the practitioners are currently utilising and tease out management principals that best suit the orthopteran assemblage, and other taxa. As production is important, we must be aware that the management recommendations must not have any impact on the productivity of the grasslands, as farmers will not sacrifice productivity to gain biodiversity.

The only two changes that I would propose to current management are, firstly, less frequent burning events, i.e. every fourth year. Fewer burning incidents will improve the vigour of the grass species allowing greater production, and reducing the formation of bare patches, allowing for the invasion of undesirable species. Species that prefer more frequent fires will be able to make use of widespread fire breaks which are burnt yearly and play an important role in preventing run away fires and resultant loss of fodder, or destruction of afforested areas. Secondly, I would try to incorporate some form of mowing regime where the grasslands are mowed in late May once the grasslands are dormant and most of the invertebrate taxa are less active and entering their over-wintering phase. This would favour the protection of over-wintering butterflies, which are usually in their most vulnerable state as larvae or pupae. Mowing is also beneficial for promoting the spread of valuable production grasses, such as, *Themeda triandra*.

Various forms of land use surround isolated grassland fragments. In this matrix grasslands interface with forestry, natural forest fragments and commercial crop production, namely maize, potatoes and pastures. These isolated fragments will require more intensive management than the extensive grasslands. The populations of invertebrates are smaller and more prone to extirpation (Steffan-Dewenter & Tschardtke, 2001). Baguette *et al.*, (2000) showed that as the size of fragments increased so the amount of emigration that was taking place of three butterfly species decreased. Emigration was in response to the lack of the correct habitat to sustain the butterfly species. As areas become smaller so management needs to be intensified to maintain ecosystem processes. Small isolated grasslands should be broken up into a mosaic of smaller units (Swengel, 2001). These smaller units should not be randomly selected, and should reflect the heterogeneous nature of the grassland fragment to be managed. For example, should their be a patch of long grass with a high forbaceous content, surrounded by short *Themeda triandra* dominated grassland, I would propose

that only half the patch be exposed to disturbance under any one management regime. This will intensify management, but will ensure that these isolated fragments are able to maintain viable populations. These smaller units need to simulate natural disturbance, and would involve rotational management. This form of management would create refugia/reservoirs of biodiversity within the grassland, which will help with the recolonization of the burnt and mowed sub units within the management matrix. Reserve areas of prairie (Panzer, 2003) can play an important role as source habitats, having a positive effect on the surrounding areas, applying to plants as well as insects (Smart *et al.*, 2002).

Some form of light intensity grazing needs to be introduced into areas, which are presently under-utilised. Grazing will create more habitat heterogeneity, and will result in the creation of a greater number of niches for colonization. In addition, grazing is useful, as it helps with the recruitment and germination of grass species. Du Toit, (1998b) suggests that a mixture of livestock units, i.e. sheep and cattle has produced a more efficient utilisation of resources than by grazing of cattle or sheep separately. This form of utilisation would mimic what happened prior to human intervention with the wild concentrate and bulk grazers feeding together.

Mowing should be undertaken on small sections of these grasslands, to create an open, short grass sward providing an area for basking, oviposition and aggregation. It is essential that the slashed grass be removed, to prevent smothering of bare areas, and impacting on soil temperatures, which may also impact on the below ground invertebrates and have an effect on overwintering eggs oviposited prior to mowing.

The benefit of mowing *per se* is questionable in the South African context, as grasshopper richness is higher in grasslands that are burnt and experience some form of utilisation. Where the grass is mowed, fire has been excluded, and I would conclude that the reason for the elevated species richness of butterflies and the elevated abundance of brachypterous endemic grasshoppers is a response to fire exclusion, not mowing. In areas where sufficient defoliation is not taking place through grazing, I would recommend that small sub-units of the grassland be mowed in order to create a defoliation event that will mimic fire, but not be as destructive.

I would propose that fire in the KwaZulu-Natal Midlands be utilised more infrequently, i.e. every four years. A reduced burning frequency will favour certain plant species, promoting flowering and re-establishment, encourage the more specialist butterfly species to recolonize these areas, and promoting increased abundance of rare and flightless grasshopper species, such as *Dirshia abbreviata*, *Qachasia fastigiata*, *Whitea alticeps*. Much of the grassland in the study area, particularly in the Midlands Mistbelt Category is *Themeda triandra* dominated, which grows to c.a. 45cm. Only once the vegetation gets taller than c.a. 90cm that the effects of light and heat exclusion negatively impact the richness of the grasshopper assemblage. Therefore biennial burning is not essential to maintain a short *Themeda triandra* dominated grass sward.

Outside formal reserves, it is necessary to maintain as much-undisturbed habitat as possible. However, this implies building in management practices that simulate natural disturbance without imposing anthropogenic disturbance that decreases habitat quality. Particular species may require appropriate management actions, but these can be chosen in a way so that both threatened specialists and more abundant generalists can benefit. Such management may involve multiple approaches (Swengel, 2001), which also need to be sensitive to other non-insect taxa. We see this with prairie management, where the fine filter specialist butterfly considerations, as well as coarse-filter total butterfly assemblage approaches, also need to cater for fire-sensitive snails (Nekola, 2002).

In the case of the rare lycaenid butterfly *Orachrysops ariadne* the fine-filter approach has priority, on the local basis (population nodes), over the coarse-filter approach, (Lu & Samways, 2002), without this prioritization this species may well go extinct. Once such highly threatened species have been considered, then a whole landscape approach can be taken. This form of management has received considerable attention in recent years with emphasis, for example, on less intensive grazing or a return to traditional methods to conserve species richness and endemism of Spanish (Verdú *et al.*, 2000) and Italian (Barbero *et al.*, 1999) dung beetles, Ukrainian (Ellingsen *et al.*, 1997) and German (Dolek & Geyer, 1997) butterflies, as well as Australian spiders (Zulka *et al.*, 1997). The complementarity approach to conservation is the only avenue to take in the way forward to prevent ever-increasing rates of extinction, especially within the invertebrate taxa.

In conclusion it must be stated that no one management type is the most appropriate or correct method for grassland management. Various factors need to be taken into consideration prior to instigating a management regime. In order to protect biodiversity in the KwaZulu-Natal Midlands context I would propose four critical factors need addressing. These factors influence the species assemblages that were studied.

1. The time between scheduled burns,
2. Management needs to be rotational, both with livestock grazing patterns and the type of defoliation that the grassland is exposed to,
3. Grazing intensity needs more vigorous assessment, and certain study sites require a reduction in animal units, and
4. Monitoring and adaptive management are the key components to ensuring the survival of rare and endemic species and bolstering biodiversity across taxa. Management should respond to changes in the target taxa monitored.

7.3. Future research

Certain questions regarding the focal taxa have been answered through the study and provide an insight into how these relatively common taxa respond to different management and environmental drivers. I would propose that further sampling and monitoring of these specific study sites takes place, with subsequent taxa included to assess the overall benefits of the suggested management recommendations on the wider cross-section of biodiversity. The additional taxa data will be able to provide an overall index of the biological integrity and grassland health for the grasslands of the KwaZulu-Natal Midlands, and thus the development of a rapid assessment technique.

ACKNOWLEDGEMENTS

I would like to acknowledge the following people for all their assistance.

Firstly, thanks must go to Michael Samways for all his effort and time that he has spent in assisting me both in terms of the structure and compilation of my PhD and in getting me funding in order to complete my PhD. I would also like to thank Steven Piper for helping facilitate the completion of the thesis.

I would like to thank the National Research Fund for the financial support that they provided in order for this study to be completed.

I would like to thank Craig Morris for all his assistance and time that he spent helping me with the statistics used to formulate my results and later discussion.

Thanks to must go to Dr. Adrian Armstrong that spent many afternoons with me identifying the grasshopper species, his assistance has been invaluable.

Sarah Pryke and Mark Brown, for aiding me with the identification of the butterfly specimens.

The support staff at the University of KwaZulu-Natal, namely, Jane Flockhart and Tanya Karalic.

Finally my parents for the financial sacrifices made to afford me the opportunity of studying for so long and there unwavering belief in my abilities. Unfortunately, dad, you didn't see the completed thesis, but this is for you.

REFERENCES

- Abbott, I., (1974). Numbers of plant, insect and land birds on nineteen remote islands in the southern hemisphere. *Biological Journal of the Linnean Society* **6**, 143-152.
- Abensperg-Traun, M., Arnold, G., Steven, D., Smith, G., Atkins, L., Viveen, J. & Gutter, M., (1997). Biodiversity indicators in contrasting vegetation types, a case study from Western Australia. *Memoirs of the Museum of Victoria* **56**, 637-641.
- Acocks, J.P.H., (1988). Veld types of South Africa. *Memoirs of the Botanical Survey of South Africa* **57**, 1-146.
- Adis, J., (1979). Problems of interpreting arthropod sampling with pitfall traps. *Zoologischer Anzeiger* **202**, 177-184.
- Andelman, S.J. & Fagan, W.F., (2000). Umbrellas and flagships: efficient conservation surrogates or expensive mistakes? *Proceedings of the National Academy of Sciences, USA* **97**, 5954-5959.
- Andersen, A.N., (1990). The use of ant communities to evaluate change in Australian terrestrial ecosystems, a review and a recipe. *Proceedings of the Ecological Society of Australia* **16**, 347-357.
- Andersen, A.N., (1999). My bioindicator or yours? Making the selection. *Journal of Insect Conservation* **3**, 61-64.
- Andersen, A.N., Hoffmann, B.D & Some, J., (2003). Ants as indicators of minesite restoration: community recovery at one of eight rehabilitation sites in central Queensland. *Ecological Management and Restoration* **4 (Supplement)**, 512-519.
- Anderson, N.L., 1964. Some relationships between grasshoppers and vegetation. *Annals of the Entomological Society of America* **57**, 736-742.

Anderson, N.L. & Wright, J.C., (1952). Grasshopper investigations on Montana rangelands. *Montana Agricultural Experiment Station Bulletin No. 486*.

Anderson, N.L. & Hastings, E., (1966). Some relationships between grasshoppers and vegetation. *Annals of the Entomological Society of America* **59**, 736-742.

Anderson, R.C., Leahy, T. & Dhillon, S., (1989). Numbers and biomass of selected insect groups on burned and unburned sand prairie. *American Midland Naturalist* **122**, 151-162.

Anderson, R.V., Tracy, C.R. & Abramsky, Z., (1979). Habitat selection in two species of short-horned grasshoppers: the role of thermal and hydric stresses. *Oecologia* **38**, 359-374.

Andrieu-Ponel, V. & Ponel, P., (1999). Human impact on Mediterranean wetland Coleoptera: an historical perspective at Tourves (Var, France). *Biodiversity and Conservation* **8**, 391-407.

Armstrong A.J. (2004). Personal Communication.

Armstrong, A.J. & van Hensbergen, H.J., (1996). The impacts of afforestation with pines on assemblages of native biota in South Africa. *South African Forestry Journal* **175**, 35-42.

Armstrong, A.J. & van Hensbergen, H.J., (1997). Evaluation of afforestable montane grasslands for wildlife conservation in the north-eastern Cape, South Africa. *Biological Conservation* **81**, 179-190.

Armstrong, A.J., van Hensbergen, H.J., Scott, D.F. & Milton, S.J., (1996). Are plantations 'inhospitable seas' around remnant native habitat within the southwestern Cape forestry areas? *South African Forestry Journal* **175**, 35-42.

Armstrong, A.J., Benn, G., Bowland, A.E., Goodman, P.S., Johnson, D.N., Maddock, A.H. & Scott-Shaw, C.R., (1998). Plantation forestry in South Africa and its impact on biodiversity. *South African Forestry Journal* **182**, 59-65.

Austin, A.D., (1999). The importance of "species" in biodiversity studies: lessons from a mega-diverse group – the parasitic Hymenoptera. In: *The other 99%: the conservation and biodiversity of invertebrates*. (Eds. Ponder, W. and Lunney, D.). Royal Zoological Society of New South Wales, Mosman, Australia.

Bailey, D.W., Dumont, B., Wallis De Vries, M.F., (1998). Utilisation of heterogeneous grasslands by domestic herbivores: theory to management. *Annals of Zootechnology* **47**, 321-333.

Bailey, R.C., Norris, R.H. & Reynoldson, T.B., (2001). Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. *Journal of North American Benthological Society* **20**, 280-286.

Baines, M., Hambler, C., Johnson, P.J., Macdonald, D.W. & Smith, H., (1998). The effects of arable field margin management on the abundance and species richness of Araneae (spiders). *Ecography* **21**, 74-86.

Báldi, A. & Kisbenedek, T., (1994). Comparative analysis of edge effect on bird and beetle communities. *Acta Zoologica Academiae Scientiarum Hungaricae* **40**, 1-14.

Báldi, A. & Kisbenedek, T., (1997). Orthopteran assemblages as indicators of grassland naturalness in Hungary. *Agriculture, Ecosystems and Environment* **66**, 121-129.

Barbero, E., Palestini, C. & Rolando, A., (1999). Dung beetle conservation: effects of habitat and resource selection (Coleoptera: Scarabaeoidea). *Journal of Insect Conservation* **3**, 75-84.

Barker, J.F., (1985). The distribution of Acridoidea (grasshoppers) in relation to overgrazing. *Botswana Notes and Records* **17**, 141-148.

Baskin, Y., (1994). Ecosystem function of biodiversity: regarding ideal levels of species richness, the data are few and unclear. *Bioscience* **44**, 657-660.

Basset, Y., Mavoungou, J.F., Mikissa, J.B., Missa, O., Miller, S.E., Kitching, R.L. & Alonso, A., (2004). Discriminatory power of different arthropod data sets for the biological monitoring of anthropogenic disturbance in tropical forests. *Biodiversity and Conservation* **13**, 709-732.

Beaudry, S., Duchesne, L.C. & Cote, B., (1997). Short-term effects of three forestry practices on carabid assemblages in a jack pine forest. *Canadian Journal for Forestry Research* **27**, 2065-2071.

Beccaloni, G.W. & Gaston, K.J., (1995). Predicting species richness of neotropical butterflies- Ithomiinae (Lepidoptera, Nymphalidae) as indicators. *Biological Conservation* **71**, 77-86.

Beckerman, A., (2000). Counterintuitive outcomes of interspecific competition between two grasshopper species along a resource gradient. *Ecology* **81**, 948-957.

Bei-Bienko, G.Y., (1970). Orthopteroid insects (Orthopteroidea) of the national park areas near Kursk and their significance as indices of local landscape. *Zhurnal Obshchei Biologii* **31**, 30-46. [In Russian, translated].

Belovsky, G.E., (2000). Grasshoppers as integral elements of grasslands. In: *Grasshoppers and Grassland Health*. (Eds. Lockwood, J.A., Latchininsky, A.V. & Sergeev, M.G.). Kluwer, Dordrecht, The Netherlands. pp. 7-29.

Belovsky, G.E. & Slade, J.B., (1993). The role of vertebrate and invertebrate predators in a grasshopper community. *Oikos* **68**, 193-201.

Belsky, A.J., (1992). Effects of grazing, competition, disturbance and fire on species composition and diversity in grassland communities. *Journal of Vegetation Science* **3**, 187-200.

Bieringer, G. & Zulka, K.P., (2003). Shading out species richness: edge effect of a pine forest on the Orthoptera (Tettigoniidae and Acrididae) assemblage of an adjacent dry grassland. *Biodiversity and Conservation* **12**, 1481-1495.

Bigalke, R.C. & Willan, K. (1984). Effects of fire regime on faunal composition and dynamics. In: *Ecological effects of fire in South African ecosystems* (Eds. Booysen, P. de V. & Tainton, N.M.) Springer-Verlag, Berlin, Germany. pp. 255-271.

Blair, J.M., (1997). Fire, N availability, and plant response in grasslands: a test of the transient maxima hypothesis. *Ecology* **78**, 2359-2368.

Blair, R.B. & Launer, A.E., (1997). Butterfly diversity and human land use: species assemblages along an urban gradient. *Biological Conservation* **80**, 113-125.

Bock, C.E. & Bock, J.H., (1991). Response of grasshoppers (Orthoptera: Acrididae) to wildfire in southeastern Arizona grassland. *American Midland Naturalist* **125**, 162-167.

Bock, C.E., Bock, J.H. & Grant, M.C., (1992). Effects of bird predation on grasshopper densities in an Arizona grassland. *Ecology* **73**, 1706-1717.

Bohac, J., (1999). Staphylinid beetles as bioindicators. *Agriculture Ecosystems and Environment* **74**, 357-372.

Bourquin, S.L., (2002). Personal Communication.

Bosch, O.J.H. & Gauch, H.R., (1991). The use of degradation gradients for the assessment and ecological interpretation of range condition. *Journal of the Grassland Society of Southern Africa* **8**, 138-146.

Boshoff, C., (1988). Does the brown locust play an important role in the nutrient recycling? In: *Proceedings of the locust symposium in Kimberley, South African Institute of Ecologists Bulletin*. (Eds. McKenzie, B. & Longridge, M.). Johannesburg, South Africa. pp. 101-117.

Bossenbroek, P.H., Kessler, A., Liem, A.S.N. & Vlijm, L., (1977). An experimental analysis of the significance of tuft-structure as a shelter for invertebrate fauna, with respect to wind velocity and temperature. *Journal of Zoology (London)* **182**, 7-16.

Bothma, J. du P., Van Rooyen, N. & Du Toit, J.G., (2002). Antelope and other small herbivores. In: *Game ranch management*. (Ed. Bothma, J. du P.) (4th Edition). Van Schaik Publishers, Pretoria, South Africa. pp. 149-175.

Boulton, A.J., (1999). An overview of river health assessment: philosophies, practice, problems and prognosis. *Freshwater Biology* **41**, 469-479.

Bourn, N.A.D. & Thomas, J.A., (2002). The challenge of conserving grassland insects at the margins of their range in Europe. *Biological Conservation* **104**, 285-292.

Boyd, N.S., Astatkie, T., Fredeen, A.H. & Martin, R.C., (2001). Rotationally stocked beef cattle responses to daily and weekly residence. *Journal of Rangeland Management* **54**, 39-43.

Bragg, T.B., (1995). The physical environment of Great Plains grasslands. In: *The changing prairie*. (Eds. Joern, A. & Keeler, K.H.). Oxford University Press, Oxford, United Kingdom. pp. 49-81.

Bridgewater, P., Walton, D.W. & Busby, J.R., (1996). Landscape diversity- creating policy. In: *Biodiversity- broadening the debate*. (Ed. Longmore, R.) ANCA Publication, Canberra, Australia. pp. 4-17.

Browde, J.A., Pedigo, L.P., De Gooyer, L.G., Higley, L.G., Wintersteen, W.K. & Zeiss, M.R., (1992). Comparison of sampling techniques for grasshoppers (Orthoptera: Acrididae) in soybean. *Journal of Economic Entomology* **85**, 2270-2274.

Brown, H.D., (1962). New and interesting grasshoppers from southern Africa – 2 (Orthoptera: Acridoidea). *Journal of the Entomological Society of Southern Africa* **25**, 3-19.

Brown, H.D., (1974). Class Insecta, Order Orthoptera, Acridoidea. In: *Status of the taxonomy of the Hexapoda of southern Africa*. (Ed. Coaton, W.G.H.). Department of Agricultural Technical Services Entomological Memoir. **38**, pp. 39-42.

Brown, J.H., 1995. *Macroecology*. Chicago University Press, Chicago, United States of America.

Brown, K.S., (1991). Conservation of neotropical environments: insects as indicators. In: *The conservation of insects and their habitats*. (Eds. Collins, N.M. & Thomas, J.A.) Academic Press, London, United Kingdom. pp. 349-404.

Brown, K.S., Jr. & Frietas, A.V.L., (2000). Atlantic forest butterflies: indicators for landscape conservation. *Biotropica* **32**, 934-956.

Bryant, J.P., Heitkonig, I., Kuropat, P. & Owen-Smith, N., (1991). Effects of severe defoliation on the long-term resistance to insect attack and on leaf chemistry in six woody species of the southern African savanna. *American Naturalist* **137**, 50-63.

Bulan, C.A. & Barrett, G.W., (1971). The effects of two acute stresses on the arthropod component of an experimental grassland ecosystem. *Ecology* **52**, 597-605.

Bullock, J.M., Hill, B.C., Dale, M.P. & Silvertown, J., (1994). An experimental study of the effects of sheep grazing on vegetation change in a species-poor grassland and the role of seedling recruitment into gaps. *Journal of Applied Ecology* **31**, 493-507.

Burd, M., (1994). Bateman's principle and plant reproduction: the role of pollen limitation in fruit and seed set. *Botanical Review* **60**, 83-139.

Cagnolo, L., Molina, S.I. & Valladares, G.R., (2002). Diversity and guild structure of insect assemblages under grazing and exclusion regimes in a montane grassland from Central Argentina. *Biodiversity and Conservation* **11**, 407-420.

Caley, M. J. & Schluter, D., (1997). The relationship between local and regional diversity. *Ecology* **78**, 70-80.

Camp, K.G.T., (1999). The bioresource groups of KwaZulu-Natal. In: *Veld in KwaZulu-Natal*. (Eds. Hardy, M.B. & Hurt, C.R.). KwaZulu-Natal Department of Agriculture. pp. 71-73.

Camp, K.G.T. & Hardy, M.B., (1999). Veld condition assessment. In: *Veld in KwaZulu-Natal*. (Eds. Hardy, M.B. & Hurt, C.R.). KwaZulu-Natal Department of Agriculture, South Africa. pp. 18-31.

Capinera, J.L. & Sechrist, T.S., (1982). Grasshopper (Acrididae)- host plant associations: response of grasshopper populations to cattle grazing. *Canadian Entomologist* **114**, 1055-1061.

Caro, T. M. & O'Doherty, G., (1999). On the use of surrogate species in conservation biology. *Conservation Biology* **13**, 805-814.

Cellier, S., (1994). Are all trees green? The forestry industry replies. *Africa – Environment and Wildlife* **2**, 79-85.

Chambers, B.Q., (1992). *Influences of burning and mowing on grasshopper (Orthoptera) assemblages in the tall grassveld of Natal*. M.Sc. thesis. University of Natal, Pietermaritzburg, South Africa.

Chambers, B.Q. & Samways, M.J., (1998). Grasshopper response to a 40-year experimental burning and mowing regime, with recommendations for invertebrate conservation management. *Biodiversity and Conservation* **7**, 985-1012.

Chappell, M.A., (1983). Metabolism and thermoregulation in desert and montane grasshoppers. *Oecologia* **56**, 126-131.

Chappell, M.A. & Whitman, D.W., (1990). Grasshopper thermoregulation. In: *Biology of grasshoppers*. (Eds. Chapman, R.F. & Joern, A.). Wiley, New York, United States of America. pp. 143-172.

Chase, J.M., (1996). Differential competitive interactions and the included niche: an experimental analysis with grasshoppers. *Oikos* **76**, 103-112.

Chase, J. M. & Leibold, M. A. (2002). Spatial scale dictates the productivity – biodiversity relationship. *Nature* **416**, 427-430.

Cherrill, A.J. & Brown, V.K., (1990). The life cycle and distribution of the wart-biter *Decticus verrucivorus* (L) (Orthoptera: Tettigoniidae) in a chalk grassland in southern England. *Biological Conservation* **53**, 125-143.

Christensen, N.L., (1997). Managing for heterogeneity and complexity on dynamic landscapes. In: *The ecological basis for conservation: heterogeneity, ecosystems and biodiversity* (Eds. Pickett, S.T.A., Ostfeld, R.S., Shachak, M. & Likens, G.E.). Chapman and Hall, New York, United States of America. pp. 167-186.

Churchill, T.B., (1997). Spiders as ecological indicators, an overview for Australia. *Memoirs of the Museum of Victoria* **56**, 331-337.

Chutter, F. M., (1994). The rapid biological assessment of streams and river water quality by means of macroinvertebrate communities in South Africa. In: *Classification of Rivers and Environmental Health Indicators*. (Ed. Uys, M.C.). Report TT 63/94, Water Research Commission, Pretoria, South Africa. pp. 217-234.

Chutter, F.M., (1998). Research on the rapid biological assessment of water quality impacts in streams and rivers. WRC Report No 422/1/98. Water Research Commission, Pretoria, South Africa.

Cincotta, R.P., Wisniewski, J. & Engelman, R., (2000). Human population in the biodiversity hotspots. *Nature* **404**, 990-992.

Claridge, M.F. & Singhrao, J.S., 1978. Diversity and altitudinal distribution of grasshoppers (Acridoidea) on a Mediterranean mountain. *Journal of Biogeography* **5**, 239-250.

- Coffin, D.P. & Lauenroth, W.K., (1988). The effects of disturbance, size and frequency on a shortgrass plant community. *Ecology* **69**, 1609-1617.
- Collinge, S.K., Prudic, K.L. & Oliver, J.C., (2003). Effects of local habitat characteristics and landscape context on grassland butterfly diversity. *Conservation Biology* **17**, 178-187.
- Collins, N.M. & Morris, M.G. (1985). *Threatened swallowtail butterflies of the world*. IUCN, Gland, Switzerland and Cambridge, UK.
- Collins, N.M. & Thomas, J.A., (1991). *The conservation of insects and their habitats*. 15th Symposium of the Royal Entomological Society of London. Academic Press, United Kingdom.
- Collins, S.L., (1987). Interactions of disturbances in tall grass prairie: a field experiment. *Ecology* **68**, 1243-1250.
- Colville, J., Picker, M.D. & Cowling, R.M., (2002). Species turnover of monkey beetles (Scarabaeidae: Hopliini) along environmental and disturbance gradients in the Namaqualand region of the succulent Karoo, South Africa. *Biodiversity and Conservation* **11**, 243-264.
- Colwell, R.K. & Coddington, J.A., (1994). Estimating terrestrial biodiversity through extrapolation. *Philosophical Transactions of the Royal Society of London Series B. Biological Sciences* **345**, 101-118.
- Committee of Scientific Issues in the Endangered Species Act (CSIESA) (1995). *Science and Endangered Species Act*. National Academy Press, Washington, D.C., United States of America.
- Costanza, R & Mageau, M., (1999). What is a healthy ecosystem? *Aquatic Ecology* **33**, 105-115.

Cracraft, J., (1992). Explaining patterns of biological diversity: integrating causation at different temporal and spatial scales. In: *Systematics, ecology, and the biodiversity crisis*. (Ed. Eldredge, N.). Columbia University Press, New York, United States of America. pp. 59-76.

Cranston, P.S. & Trueman, J.W.H., (1997). 'Indicator' taxa in invertebrate biodiversity assessment. *Memoirs of the Museum of Victoria* **56**, 264-267.

Crawley, M.J., (1983). *Herbivory: the dynamics of animal-plant interactions*. Blackwell Scientific Publications, Oxford, United Kingdom.

Cully, J.F.Jr., (1999). Lone star tick abundance, fire, and bison grazing tall-grass prairie. *Journal of Range Management* **52**, 139-144.

Currie, D.J., (1991). Energy and large-scale patterns of animal and plant species richness. *The American Naturalist* **137**, 27-49.

Daubenmire, R., (1968). Ecology of fire in grasslands. *Advanced Ecological Research* **5**, 209-266.

Davidowitz, G. & Rosenzweig, M.L., (1998). The latitudinal gradient of species diversity among North American grasshoppers (Acrididae) within a single habitat: a test of the spatial heterogeneity hypothesis. *Journal of Biogeography* **25**, p 553-560.

Dawson, B.L., (1991). South African road reserves: valuable conservation areas? In: *The role of corridors*. (Eds. Saunders, D.A. & Hobbs, R.J.). Surrey Beatty and Sons Publishers, Chipping Norton, United Kingdom.

Day, T.A. & Detling, J.K., (1990). Grassland patch dynamics and herbivore grazing preference following urine deposition. *Ecology* **71**, 180-188.

De Maynadier, P. & Hunter, M. L., (1994). Keystone support. *BioScience* **44**, 2.

Debinski, D.M., Ray, C. & Saveraid, E.H., (2001). Species diversity and the scale of the landscape mosaic: do scales of movement and patch size affect diversity? *Biological Conservation* **98**, 179-190.

Dempster, J.P., (1963). The population dynamics of grasshoppers and locusts. *Biological Reviews* **38**, 490-529.

Dennis R.L.H., (1993). *Butterflies and Climate Change*. Manchester University Press, Manchester, United Kingdom.

Deutschländer, M. S. & Bredenkamp, G. J., (1999). Importance of vegetation analysis in the conservation management of the endangered butterfly *Aloeides dentatis dentatis* (Swierstra) (Lepidoptera, Lycaenidae). *Koedoe* **42**, 1-12.

Devoka, D. & Schmidt, G.H., (2000). Accumulation of heavy-metals in food plants and grasshoppers from the Taigetos Mountains, Greece. *Agriculture, Ecosystems and Environment* **78**, 85-91.

Dickens C.W.S. & Graham P.M., (2002). The South African scoring system (SASS) version 5 rapid bioassessment method for rivers. *African Journal of Aquatic Science* **27**, 1-10.

Didham, R.K., Ghazoul, J., Stork, N.E. & Davis, A.J., (1996). Insects in fragmented forests: a functional approach. *Trend in Ecology and Evolution* **11**, 255-260.

Dirsh, V.M., (1965). *The African genera of Acridoidea*. Anti-Locust Research Centre and Cambridge University Press, Cambridge, United Kingdom.

Dolek, M. & Geyer, A., (1997). Influence of management on butterflies of rare grassland ecosystems in Germany. *Journal of Insect Conservation* **1**, 125-130.

Dougill, A. & Trodd, N., (1999). Monitoring and modeling open savannas using multisource information: analysis of Kalahari studies. *Global Ecology and Biogeography* **8**, 211-221.

Dourojeanni, M.J., (1990). Entomology and biodiversity conservation in Latin America. *American Naturalist* **36**, 88-93.

Duffey, E., (1968). Ecological studies on the large copper butterfly *Lycaena dispar* Haw. *batavus* Obth. at Woodwalton Fen National Nature Reserve, Huntingdonshire. *Journal of Applied Ecology* **5**, 69-96.

Duffey, E., Morris, M.G., Sheail, J., Ward, L.K., Wells, D.A. & Wells, T.C.E., (1974). *Grassland ecology and wildlife management*. Chapman and Hall, London, United Kingdom.

Dufrêne, M. & Legendre, P., (1997). Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs* **67**, 345-366.

Dukas, R. & Bernays, E.A., (2000). Learning improves growth rate in grasshoppers. *Physiological and Biochemical Zoology* **73**, 66-76.

Du Toit, P.C.V., (1998b). Effects of grazing and resting treatments on animal performance and vegetation condition in the false upper Karoo at the Grootfontein Agricultural Development Institute, Eastern Cape. *South African Journal of Science* **94**, 507-512.

Erhardt, A., (1985). Diurnal Lepidoptera: sensitive indicators of cultivated and abandoned grassland. *Journal of Applied Ecology* **22**, 849-862.

Erhlich, P.R., Breedlove, D.E., Brussard, P.F. & Sharp, M., (1972). Weather and the regulation of sub alpine populations. *Ecology* **53**, 243-247.

Ehrlich, P.R. & Murphy, D. D., (1987). Conservation lessons from long-term studies of checkerspot butterflies. *Conservation Biology* **1**, 122-131.

Ericson, D., (1979). The interpretation of pitfall catches on *Pterostichus cupreus* and *Pt. melanarius* (Coleoptera: Carabidae) in cereal fields. *Pedobiologia* **19**, 320-328.

- Elligsen, H., Beinlich, B. & Plachter, H., (1997). Effects of large-scale cattle grazing on populations of *Coenonympha glycerion* and *Lesionmata megera* (Lepidoptera: Satyridae). *Journal of Insect Conservation* **1**, 13-23.
- Evans, E.W., (1984). Fire as a natural disturbance to grasshopper assemblages of tall grass prairie. *Oikos* **43**, 9-16.
- Evans, E.W., (1988a). Grasshopper (Insecta: Orthoptera: Acrididae) assemblages of tall grass prairie: influences of fire frequency, topography and vegetation. *Canadian Journal of Zoology* **66**, 1495-1501.
- Evans, E.W., (1988b). Community dynamics of prairie grasshoppers subjected to periodic fire: predictable trajectories or random walks in time? *Oikos* **52**, 283-289.
- Everson, T.M., Clarke, G.P.Y. & Everson, C.S., (1990). Precision in monitoring plant species composition in montane grasslands. *Vegetatio* **88**, 135-141.
- Feber, R.E., Smith, H. & Macdonald, D.W., (1996). The effects of management of uncropped edges of arable fields on butterfly abundance. *Journal of Applied Ecology* **33**, 1191 – 1205.
- Field, L.F., (2002). *Consequences of habitat fragmentation for the pollination of wildflowers in moist upland grasslands of KwaZulu-Natal*. M.Sc. thesis University of Natal, Pietermaritzburg, South Africa.
- Fielding, D.J. & Brusven, M.A., (1993a). Grasshopper (Orthoptera: Acrididae) community composition and ecological disturbance on southern Idaho rangeland. *Environmental Entomology* **22**, 71-81.
- Fielding, D.J. & Brusven, M.A., (1993b). Spatial analysis of grasshopper density and ecological disturbance on southern Idaho rangeland. *Agriculture, Ecosystems and Environment* **43**, 31-47.

- Fielding, D.J. & Brusven, M.A., (1995). Grasshopper densities on grazed and ungrazed rangeland under drought conditions in southern Idaho. *Great Basin Naturalist* **55**, 352-358.
- Fielding, D.J., Brusven, M.A., Shafii, B. & Price, W.J., (2001). Spatial heterogeneity of low-density populations of *Melanoplus sanguinipes* (Orthoptera: Acrididae) associated with grazing and vegetation treatments. *Canadian Entomologist* **133**, 843-855.
- Fishman, J., Fakhruzzman, K., Cros, B. & Hganga, D., (1991). Identification of widespread pollution in the southern hemisphere deduced from satellite analyses. *Science* **252**, 1693-1696.
- Fogden, M. & Fogden, P., (1974). *Animals and their colours*. Crown, New York, United States of America.
- Force, D., (1981). Postfire insect succession in southern California chaparral. *The American Naturalist* **117**, pp 575-582.
- Forestry Council, (1993). *South African forestry facts*. Forestry Council, Rivonia, South Africa.
- Forestry Council, (2004). *South African forestry facts*. Forestry Council, Rivonia, South Africa.
- Foord, S.H., Ferguson, J.W.H. & van Jaarsveld, A.S., (2002). Endemicity of afro-montane grasshopper assemblages: implications for grassland management. *African Journal of Ecology* **40**, 318-327.
- Frank, D.A., McNaughton, S.J. & Tracy, B.F., (1998). The ecology of the earth's grazing ecosystems. *BioScience* **48**, 513-521.
- Franklin, J. F., (1993). Preserving biodiversity: species, ecosystems or landscapes? *Ecological Applications* **3**, 202-205.

Freeman, C.C., (1998). The flora of Konza Prairie: a historical review and contemporary patterns. In: *Grassland dynamics: long-term ecological research in tallgrass prairie*. (Eds. Knapp, A.K., Briggs, J.M., Hartnett, D.C. & Collins, S.L.). Oxford University Press, New York, United States of America. pp. 69-80.

Frost, P.G.H., (1984). The responses and survival of organisms in fire prone environments. In: *Ecological effects of fire in South African ecosystems* (Eds. Booysen, P. de V. & Tainton, N.M.) Springer-Verlag, Berlin, Germany. pp. 273-309.

Fry, R. & Lonsdale, D., 1991. *Habitat conservation for insects – a neglected green issue*. Amateur Entomologists' Society, Middlesex, United Kingdom.

Fuhlendorf, S.D. & Smeins, F.E., (1997). Long-term vegetation dynamics mediated by herbivores, weather and fire in a *Juniperus-Quercus* savanna. *Journal Vegetation Science* **8**, 819-818.

Fuhlendorf, S.D. & Engle, D.M., (2001). Restoring heterogeneity on rangelands: ecosystem management based on grazing patterns. *Bioscience* **51**, 625-632.

Fuhlendorf, S.D., Woodward, A.J.W., Leslie, D.M.Jr. & Shackford, J., (2002). Multi-scale effects of habitat loss and fragmentation on lesser prairie-chicken populations of the US southern Great Plains. *Landscape Ecology* **17**, 617-628.

Fynn, R.W.S., Haynes, R.J. & O'Connor, T.G., (2003). Burning causes long-term changes in the soil organic matter content of a South African grassland. *Soil Biology and Biochemistry* **35**, 677-687.

Fynn, R.W.S., Morris, C.D. & Edwards, T.J., (2005). Long-term compositional responses of a South African mesic grassland to various combinations of burning and mowing. *Applied Vegetation Science*. In Press.

Gall, L.F., (1985). Measuring the size of lepidopteran populations. *Journal of Research on the Lepidoptera* **24**, 97-116.

Gandar, M.V., (1979). Studies on the insects of the herbaceous layer of the Nylsvley savanna. *Final Report to the National Program for Environmental Sciences*. **69**.

Gandar, M.V., (1980). Short-term effects of the exclusion of large mammals and insects in broad leaf savanna. *South African Journal of Science* **76**, 29-31.

Gandar, M.V., (1982a). The dynamics and trophic ecology of grasshoppers (Acridoidea) in a South African savanna. *Oecologia* **54**, 370-378.

Gandar, M.V., (1982b). Description of a fire and its effects in the Nylsvley nature reserve: a synthesis report. *South African National Scientific Programmes Report No. 74. Council for Scientific and Industrial Research*.

Gandar, M.V., (1983). Ecological notes and annotated checklist of the grasshoppers (Orthoptera: Acrididae) of the savanna ecosystem project study area, Nylsvley. *South African National Scientific Programmes Report No. 74. Council for Scientific and Industrial Research*.

Gangwere, S.K., Muralirangan, M.C. & Muralirangan, M., (1997). *The bionomics of grasshoppers, katydids and their kin*. CAB International, Wallingford, United Kingdom.

Gaston, K.J. & Blackburn, T.M., (1995). Mapping biodiversity using surrogates for species richness: macro-scales and New World birds. *Proceedings of the Royal Society of London Series B. Biological Sciences* **262**, 335-341.

Gaston, K.J. & Blackburn, T.M., (1996). The tropics as a museum of biological diversity: an analysis of the New World avifauna. *Proceedings of the Royal Society of London Series B. Biological Sciences* **263**, 63-68.

Gaston, K.J., New, T.R. & Samways, M.J., (Eds.), 1993. *Perspectives on insect conservation*. Intercept, Andover, United Kingdom.

- Gebeyehu, S. & Samways, M.J., (2002). Grasshopper assemblage response to a restored national park (Mountain Zebra National Park, South Africa). *Biodiversity and Conservation* **11**, 283-304.
- Gebeyehu, S. & Samways, M.J., (2003). Responses of grasshopper assemblages to long-term grazing management in a semi-arid African savanna. *Agriculture, Ecosystems and Environment* **95**, 613-622.
- Germishuizen, G. & Meyer, N.L., (Eds.), (2003). Plants of southern Africa: an annotated checklist. *Strelitzia 14*. National Botanical Institute, Pretoria, South Africa.
- Ghazoul, J. & Hellier, A., (2000). Setting critical limits to ecological indicators of sustainable tropical forestry. *International Forestry Review* **2**, 243-253.
- Gibson, C.W.D., Hambler, C. & Brown, V.K., (1992). Change in spider (Araneae) assemblages in relation to succession and grazing management. *Journal of Applied Ecology* **29**, 132-142.
- Gibson, D.J. & Hulbert, L.C., (1987). Effects of fire, topography, and year-to-year climatic variation on species composition in tallgrass prairie. *Vegetatio* **72**, 175-185.
- Giller, S., (1996). The diversity of soil communities, the 'poor man's tropical rain forest. *Biodiversity and Conservation* **5**, 135-168.
- Gillon, Y., (1983). The invertebrates of the grass layer. In: *Ecosystems of the world 13: tropical savannas*. (Ed. Boulière, F.). Elsevier, Amsterdam, The Netherlands. pp. 289-311.
- Goldsmith, F.B., (1991). *Monitoring for conservation and ecology*. Chapman and Hall, London, United Kingdom.

Grant, S.A., Torvell, L., Sim, E.M., Small, J.L. & Armstrong, R.H., (1996). Controlled grazing studies on *Nardus* grassland: effects of between tussock sward height and species of grazer on *Nardus* utilisation and floristic composition in two fields in Scotland. *Journal of Applied Ecology* **33**, 1053-1065.

Greatorrex-Davis, J.N., Sparks, T.H. & Hall, M.L., (1994). The response of Heteroptera and Coleoptera species to shade and aspect in rides of coniferised lowland. *Biological Conservation* **12**, 1099-1111.

Green, S.V., (1998). The taxonomic impediment in orthopteran research and conservation. *Journal of Insect Conservation* **2**, 151-159.

Green, S.V., (1999). South African Acridoidea (Orthoptera) – a review of African Entomology. Unpublished.

Greenslade, P. (1964). Pitfall trapping as a method of studying populations of Carabidae (Coleoptera). *Journal of Animal Ecology* **33**, 301-310.

Greenslade, P., (1993). Australian native steppe-type landscapes: neglected areas for invertebrate conservation in Australia. In: *Perspectives on Insect Conservation*. (Eds. Gaston, K.J., New, T.R. & Samways, M.J.) Intercept, Andover, United Kingdom. pp. 51-73.

Greenslade, P. & Greenslade, P.J.M., (1987). Ecological strategies in Collembola, a new approach to the use of terrestrial invertebrates in environmental assessment. In: *Ecological strategies in Collembola, a new approach to the use of terrestrial invertebrates in environmental assessment*. (Ed. Striganova, B.R.). Nauka, Moscow, Russia. pp. 245-252.

Groom, M.J. & Schumaker, N., (1993). Evaluating landscape change: patterns of worldwide deforestation and local fragmentation. In: *Biotic interactions and global change*. (Eds. Kareiva, P.M., Kingsolver, J.G. & Huey, R.B.). Sinauer Associates Incorporated, Massachusetts, United States of America. pp. 9-23.

- Guérol, F., (2000). Influence of taxonomic determination level on several community indices. *Water Research* **34**, 487-492.
- Gutzwiller, K.J. 2002. Applying landscape ecology in biological conservation: principles, constraints, and prospects. In: *Applying landscape ecology in biological conservation*. (Ed. Gutzwiller, K.J.). Springer-Verlag Incorporated, New York, United States of America. pp. 481-495
- Haes, E.C.M., Cherrill, A.J. & Brown, V.K, (1990). Meteorological correlates of wart-biter (Orthoptera: Tettigoniidae) abundance. *The Entomologist* **109**, 93-99.
- Hall, M., (1984) Man's historical and traditional use of fire in southern Africa. In: *Ecological effects of fire in South African ecosystems*. (Eds. Booysen, P. de V. & Tainton, N.M.). Springer-Verlag, Berlin, Germany. pp. 39-54.
- Hamer, K.C., Hill, J.K., Lace, L.A. & Langan, A.M., (1997). Ecological and biogeographical effects of forest disturbance on tropical butterflies in Sumba, Indonesia. *Journal of Biogeography* **24**, 67-75.
- Hammond, P., (1994). Practical approaches to the estimation of the extent of biodiversity in speciose groups. *Philosophical Transactions of the Royal Society of London B* **345**, 119-136.
- Hannah, L., Lohse, D., Hutchinson, C., Carr, J.L. & Lanckerani, A., (1994). A preliminary inventory of human disturbance of world ecosystems. *Ambio* **23**, 246-250.
- Hanski, I., (1994). A practical model of metapopulation dynamics. *Journal of Applied Ecology* **63**, 151-162.
- Harding, P.T., Asher, J. & Yates, T.J., (1995). Butterfly monitoring 1- recording the changes. In: *Ecology and conservation of butterflies*. (Ed. Pullin, A.S.). Chapman and Hall, London, United Kingdom. pp. 3-22.

- Hardy, M.B., (1999). Bioresource groups 8 and 9: moist and dry highland sourveld. In: *Veld in KwaZulu-Natal*. (Eds. Hardy, M.B. & Hurt, C.R.). KwaZulu-Natal Department of Agriculture. pp. 84-87.
- Haslett, J.R., (2001). Biodiversity and conservation of Diptera in heterogeneous land mosaics: a fly's eye view. *Journal of Insect Conservation* **5**, 71-75.
- Hawksworth, D.L. & Kalin-Arroyo, M.T., (Eds.), (1995). Magnitude and distribution of biodiversity. In: *United Nations environment program*. Cambridge University Press, Cambridge, United Kingdom. pp. 107-193.
- Haysom, K.A. & Coulson, J.C., (1998). The Lepidoptera fauna associated with *Calluna vulgaris*: effects of plant architecture on abundance and diversity. *Ecological Entomology* **23**, 377-385.
- Henning, G.A., Henning, S.F., Joannou, J.G. & Woodhall, S.E., (1997). *Living butterflies of southern Africa (Volume 1)*. Umdaus Press, Hatfield, South Africa.
- Hill, J.K., Kramer, K.C., Lace, L.A. & Banham, W.M.T., (1995). Effects of selective logging on tropical forest butterflies on Buru, Indonesia. *Journal of Applied Ecology* **32**, 754-760.
- Hill, P. & Twist, C., (1998). *Butterflies and Dragonflies: a site guide*. 2nd edition. Arlequin, Chelmsford, United Kingdom.
- Hilton-Taylor, C., (compiler), (2000). *2000 IUCN Red Lists of Threatened Species*. IUCN, Gland, Switzerland and Cambridge, United Kingdom.
- Holechek, J.L., Pieper, R.D. & Herbel, C.H., (1998). Range management principles and practices: upper Saddle River (New Jersey). Prentice Hall, United States of America.
- Holl, K.D., (1995). Nectar resources and their influence on butterfly communities on reclaimed coal surface mines. *Restoration Ecology* **3**, 76-85.

Holloway, J.D. and Stork, N.E., (1991). The dimensions of biodiversity: the use of invertebrates as indicators of human impact. In: *The biodiversity of microorganisms and invertebrates: its role in sustainable agriculture*. (Ed. Hawksworth, D.L.). CAB International, Wallingford, United Kingdom. pp. 37-62.

Holmes, N.D., Smith, D.S. & Johnston, A., (1979). Effect of grazing cattle on the abundance of grasshoppers on fescue grassland. *Journal of Range Management* **32**, 310-311.

Honek, A., (1988). The effect of crop density and microclimate on pitfall trap catches of Carabidae, Staphylinidae (Coleoptera) and Lycosidae (Araneae) in cereal fields. *Pedobiologia* **32**, 233-242.

Horwitz, P., Recher, H. & Majer, J., (1999). Putting invertebrates on the agenda: political and bureaucratic challenges. In: *The other 99%: the conservation of biodiversity of invertebrates*. (Eds. Ponder, W. & Lunney, D.). The Royal Zoological Society of New South Wales, Mosman, Australia. pp. 398-406.

Hunter, M.L.Jr., (2000b). Refining normative concepts in conservation. *Conservation Biology* **14**, 573-574.

Ibrahim, K.M., (2001). Plague dynamics and population genetics of the desert locust: can turnover during recession maintain population genetic structure? *Molecular Ecology* **10**, 581-591.

Ingham, D. S. & Samways, M. J., (1996). Application of fragmentation and variegation models to epigaeic invertebrates in South Africa. *Conservation Biology* **10**, 1353-1358.

Ingrisch, S., (1986). The plurennial lifecycles of the European Tettigoniidae (Insecta: Orthoptera). *Oecologia* **70**, 606-630.

Isely, F.B., (1938). The relations of Texas Acrididae to plants and soil. *Ecological Monographs* **8**, 551-604.

IUCN/UNEP/WWF (1991) *Caring for the Earth. A Strategy for Sustainable Living*. Gland, Switzerland.

Jaganyi, J.N.U., (1998). *Response of carabid and cicindelid beetles to various types of landscape disturbances*. PhD thesis. University of Natal, Pietermaritzburg, South Africa.

Janzen, D.H., (1987). Insect diversity in a Costa Rican dry forest: why keep it and how? *Biological Journal of the Linnaean Society* **30**, 343-356.

Jepson-Innes, K. & Bock, C.E., (1989). Response of grasshoppers (Orthoptera: Acrididae) to livestock grazing in southeastern Arizona: differences between seasons and subfamilies. *Oecologia* **78**, 430-431.

Joern, A., (1979). Resource utilisation and community structure in assemblages of arid grassland grasshoppers (Orthoptera: Acrididae). *Transactions of the American Entomological Society* **105**, 253-300.

Joern, A., (1982). Vegetation structure and microhabitat selection in grasshoppers (Orthoptera: Acrididae). *Southwestern Naturalist* **27**, 197-209.

Joern, A., (1983). Small scale displacements of grasshoppers (Orthoptera: Acrididae) within arid grasslands. *Journal of the Kansas Entomological Society* **56**, 131-139.

Joern, A. & Gaines, S.B., (1990). Population dynamics and regulation in grasshoppers. In: *Biology of grasshoppers*. (Eds. Joern, A. & Chapman, R.F.). Wiley and Sons, New York, United States of America. pp. 415-482.

Johns, M., (1993). Are all trees green? A spotlight on forestry. *Africa – Environment and Wildlife* **1**, 77-85.

Johnsen, P., (1985). Suborder Caelifera. In: *Insects of southern Africa*. (Eds. Scholtz, C.H. & Holm, E.). Butterworths, Durban, South Africa. pp. 86-95.

Johnsen, P., (1987). The status of South African Acridoidea s.i. (Orthoptera: Caelifera). In: *Evolutionary biology of orthopteroid insects*. (Ed. Baccetti, B.). Ellis Horwood, Chichester, United Kingdom. pp. 293-295.

Johnson, S., (2005). Personal Communication.

Johnson, R.A. & Wichern, D.W., 1982. *Applied multivariate statistical analyses*. Prentice-Hall, Englewood Cliffs, New Jersey, United States of America.

Jones, D.T. & Eggleton, P., (2000). Sampling termite assemblages in tropical forests: testing a rapid biodiversity assessment protocol. *Journal of Applied Ecology* **37**, 191–203.

Joy, J. & Pullin, A.S., (1999) Field studies on flooding and survival of overwintering large heath butterfly *Coenonympha tullia* larvae on Fennis and Whixall Mosses in Shropshire and Wrexham, United Kingdom. *Ecological Entomology* **24**, 426-431.

Karieva, P.M., (1993). Influence of vegetation texture on herbivore populations: resource concentrations and movement. In: *Variable plants and herbivores in natural and managed ecosystems*. (Eds. Denno, R.F. & McClure, M.S.). Academic Press, New York, United States of America. pp. 259-289.

Kati, V., De Villiers, P., Dufrêne, M., Legakis, A., Vokou, D. & Lebrun, P., (2004). Testing the value of six taxonomic groups as biodiversity indicators at a local scale. *Conservation Biology* **18**, 667-675.

Kearns, C.A., Inouye, D.W. & Waser, N.M., (1998). Endangered mutualisms: the conservation of plant-pollinator interactions. *Annual Review of Ecology and Systematics* **29**, 83-112.

Kellert, S.R. (1986). Social and perceptual factors in the preservation of animal species. In: *The preservation of species*. (Ed. Norton, B.G.). Princeton University Press, Princeton, United States of America. pp. 50-73.

- Kells, A. R. & Goulson, D. (2003). Preferred nesting sites of bumblebee queens (Hymenoptera: Apidae) in agroecosystems in the UK. *Biological Conservation* **109**, 165-174.
- Kemp, W.P., (1992). Rangeland grasshopper (Orthoptera: Acrididae) community structure. *Environmental Entomology* **21**, 461-470.
- Kemp, W.P., Harvey, S.J. & O'Neill, K.M., (1990). Patterns of vegetation and grasshopper community composition. *Oecologia* **83**, 299-308.
- Kennedy, A.D., (1994). Simulated climate change: a field manipulation study of polar microarthropod community response to global warming. *Ecography* **17**, 131-140.
- Kent, M. & Coker, P., (1992). *Vegetation description and analysis: a practical approach*. Belhaven Press, London, United Kingdom.
- Kindvall, O., (1996). Habitat heterogeneity and survival in a bush cricket metapopulation. *Ecology* **77**, 207-214.
- King, K.L. & Hutchinson, K.J., (1976). The effects of sheep stocking intensity on the abundance and distribution of mesofauna in pastures. *Journal of Applied Ecology* **13**, 41-56.
- Kirby, P., (1992). *Habitat management for invertebrates: a practical handbook*. The Royal Society for the Protection of Birds, Bedfordshire, United Kingdom.
- Kisbenedek, T., (1995). The effects of sheep grazing on the community structure of grasshoppers (Orthoptera). *Folia Entomologica Hungarica* **56**, 45-46.
- Kitahara, M. & Sei, K., (2001). A comparison of the diversity and structure of butterfly communities in semi-natural and human-modified grassland habitats at the foot of Mt. Fuji, central Japan. *Biodiversity and Conservation* **10**, 331-351.

- Knapp, A.K., Blair, J.M., Briggs, J.M., Collins, S.L., Hartnett, D.C., Johnson, L.C. & Towne, E.G., (1999). The keystone role of bison in North American tallgrass prairie. *BioScience* **49**, 39-50.
- Kolasa, J. & Pickett, S.T.A., (1991). *Ecological Heterogeneity*. Springer-Verlag, New York, United States of America.
- Kotliar, N.B. & Wiens, J.A., (1990). Multiple scales of patchiness and patch structure: a hierarchical framework for the study of heterogeneity. *Oikos* **59**, 253-260.
- Kotze, D., (2004). Personal Communication.
- Kotze, D.J. & Samways, M.J., (2001). No general edge effects for invertebrates at afro-montane forest/grassland ecotones. *Biodiversity and Conservation* **10**, 443-466.
- Kremen, C., (1990). Identifying indicator taxa for conservation planning: a study of rain forest butterflies in Madagascar. *Bulletin of the Ecological Society of America* **71**, 220.
- Kremen, C., (1992). Assessing the indicator properties of species assemblages for natural areas monitoring. *Ecological Applications* **2**, 203-217.
- Kremen, C., (1994). Biological inventory using target taxa: a case study of butterflies of Madagascar. *Ecological Applications* **4**, 407-422.
- Kruess, A. & Tscharnke, T., (2002). Grazing intensity and the diversity of grasshoppers, and trap-nesting bees and wasps. *Conservation Biology* **16**, 1570-1580.
- Kruger, F.J., (1984). Effects of fire on vegetation structure and dynamics. In: *Ecological effects of fire in South African ecosystems*. (Eds. Booyesen, P. de V. & Tainton, N.M.). Springer-Verlag, Berlin, Germany. pp. 219-243.
- Kruger, F.J., Everard, D.A. & Manders, P.T., (1995). Forestry and environmentally sustainable development in South Africa. *Commonwealth Forestry Review* **74**, 71-75.

Lactin, D.J. & Johnson, D.L., (1997). Response of body temperature to solar radiation in restrained nymphal migratory grasshoppers (Orthoptera: Acrididae): influences of orientation and body size. *Physiological Entomology* **22**, 131-139.

Lactin, D.J. & Johnson, D.L., (1998). Convective heat loss and change in body temperature of grasshopper and locust nymphs: relative importance of wind speed, insect size and insect orientation. *Journal of Thermal Biology* **23**, 5-13.

Landsberg, J., O'Connor, T. & Freudenberger, D., (1999). The impacts of livestock grazing on biodiversity in natural ecosystems. In: *Nutritional ecology of herbivores*. (Eds. Jung, H.G. & Fahley, G.C.Jr.). *Proceedings of the 5th International Symposium on the nutrition of herbivores*. American Society of Animal Science, Illinois, United States of America. pp. 752-777.

Lamotte, M., (1975). The structure and function of a tropical savanna system. In: *Tropical ecology systems. Trends in terrestrial and tropical research*. (Eds. Golley, F.B. & Medino, E.) Springer-Verlag, Berlin, Germany. pp. 179-222.

Latchininsky, A.V., (1998). Moroccan locust *Dociostaurus maroccanus* (Thunberg, 1985): a faunistic rarity or an important economic pest? *Journal of Insect Conservation* **2**, 167-178.

Launer, A.E. & Murphy, D.D., (1994). Umbrella species and the conservation of habitat fragments: a case of threatened butterfly and vanishing grassland ecosystem. *Biological Conservation* **69**, 145-153.

Lawes, M.J., Kotze, D.J., Bourquin, S.L. & Morris, C.D., (2005). Epigaeic invertebrates as potential ecological indicators of Afromontane forest condition in South Africa. *Biotropica* **37**, 109-118.

Lawton, J.H., Bignell, D.E., Bolton, B., Bloemers, G.F., Eggleton, P., Hammond, P.M., Hodda, M., Holt, R.D., Larson, T.B., Mawdsley, N.A., Stork, N.E., Srivastava, D.S., & Watt, A.D., (1998). Biodiversity inventories, indicator taxa and the effects of habitat modification in tropical forest. *Nature* **391**, 72-76.

- Legg, D.E., Lockwood, J.A. & Brewer, M.J., (1996). Variability in rangeland grasshopper (Orthoptera: Acrididae) counts when using the standard, visualized-sampling method. *Journal of Economic Entomology* **89**, 1143-1150.
- Lenat, D.R. & Resh, V.H., (2001). Taxonomy and stream ecology – the benefits of genus- and species-level identification. *Journal of North American Benthological Society* **20**, 287-298.
- Leopold, A., (1953). *Round River*. Oxford University Press, New York, USA.
- Leps, J. & Spitzer, K., (1990). Ecological determinants of butterfly communities (Lepidoptera: Papilionidae) in the Tam Dao Mountains. *Acta Entomologica Bohemoslovaca* **87**, 182-194.
- Lindenmayer, D.B., Margules, C.R. & Botkin, D.B., (1999). Indicators of biodiversity for ecologically sustainable forest management. *Conservation Biology* **14**, 941-950.
- Lockwood, J.A., (1993). Environmental issues involved in biological control of rangeland grasshoppers (Orthoptera: Acrididae) with exotic agents. *Environmental Entomology* **22**, 503-518.
- Lockwood, J.A., (1997). Rangeland grasshopper ecology. In: *The bionomics of grasshoppers, katydids and their kin*. (Eds. Gangwere, S.K., Muralirangan, M.C. & Muralirangan, M.). CAB International, Wallingford, United Kingdom. pp. 83-101.
- Lockwood, J.A., Latchininsky, A.V. & Sergeev, M.G., (2000). *Grasshoppers and grasshopper health: managing grasshopper outbreaks without risking environmental disaster*. Kluwer Academic Publishers, Dordrecht, The Netherlands.
- Longino, J.T., & Colwell, R.K., (1997). Biodiversity assessment using structured inventory: capturing the ant fauna of a lowland tropical rainforest. *Ecological Applications* **7**, 1263-1277.

Lu, S.S., (2002). *Biology and conservation of the threatened Karkloof blue butterfly Orachrysops ariadne (Butler) (Lepidoptera: Lycaenidae)*. PhD thesis. University of Natal, Pietermaritzburg, South Africa.

Lu, S.S. & Samways, M.J., (2002). Conservation management recommendations for the Karkloof blue butterfly, *Orachrysops ariadne* (Lepidoptera: Lycaenidae). *African Entomology* **10**, 149-159.

Luff, M.L., (1975). Some features influencing the efficiency of pitfall traps. *Oecologia (Berlin)* **19**, 345-357.

Luff, M.L., (1982). Population dynamics of Carabidae. *Annals of Applied Biology* **101**, 164-170.

Luff, M.L., (1986). Aggregation of some Carabidae in pitfall traps. In: *Carabid beetles, their adaptations and dynamics*. (Eds. Den Boer, P.J., Luff, M.L., Mossakowski, D. & Weber, F.). Gustav Fischer, Stuttgart, Germany.

Lunt, I.D. & Morgan, J.W., (2002). The role of fire regimes in temperate lowland grassland in southeastern Australia. In: *Flammable Australia, the fire regimes and biodiversity of a continent*. (Eds. Bradstock, R.A., Williams, J.E. & Gill, A.M.). Cambridge University Press, Cambridge, United Kingdom. pp. 177-196.

Macdonald, I.A.W., (1989). Mans role in changing the face of southern Africa. In: *Biotic diversity in southern Africa* (Ed. Huntley, B.J.). Oxford University Press, Cape Town, South Africa. pp. 51-77.

Majer, L., (1980). Report on the study of invertebrates in relation to the Kojonup nature reserve fire management plan. *WAIT Biology Department, Bulletin No. 2*.

Maliha, S.N., Walter, G.W., Justin, V.Z. & Kris, H., 2000. Ant (Hymenoptera: Formicidae) responses to environmental stressors in the northern Chihuahuan desert. *Environmental Entomology* **29**, 200-206.

MacArthur, R.H., (1972). *Geographical ecology*. Harper and Row, New York, United States of America.

MacArthur, R.H. & Wilson, E.O., (1967). *The theory of island biogeography*. Princeton University Press, New Jersey, United States of America.

MacFarlane, D., (2002). Personal Communication.

Magurran, A.E., (1988). *Ecological diversity and its measurement*. Princeton University Press, Princeton, United States of America.

Mani, M.S., 1962. *Introduction to high altitude entomology: insect life above the timberline in northwest Himalaya*. Methuen, London, United Kingdom.

Manzano, M.G. & Navar, J., (2000). Processes of desertification by goats overgrazing in the Tamaulipan thornscrub (matorral) in north-eastern Mexico. *Journal of Arid Environments* **44**, 1-17.

Marshall, J.A. & Haes, E.C.M., (1988). *Grasshoppers and allied insects of Great Britain and Ireland*. Harley Books, Colchester, United Kingdom.

Mawdsley, N.A. & Stork, N.E., (1995). Species extinctions in insects: ecological and biogeographical considerations. In: *Insects in a Changing Environment*. (Ed. Harrington, R. & Stork, N.E.). Academic Press, London, United Kingdom. pp. 321-369.

McCabe, T.L., (1981). The Dakota skipper, *Hesperia dacotae* (Skinner): range and biology, with special reference to North Dakota. *Journal of Lepidopterists' Society* **35**, 179-193.

McCune, B. & Grace, J.B., (2002). *Analysis of ecological communities*. MJM Software Design, Oregon, United States of America.

McGeoch, M.A., (1998). The selection, testing and application of terrestrial insects as bioindicators. *Biological Review* **73**, 181-201.

McIntyre, N.E. & Wiens, J.A., (1999). How does habitat patch size affect animal movement? An experiment with darkling beetles. *Ecology* **80**, 2261-2270.

Meek, B., Loxton, D., Sparks, T., Pywell, R., Pickett, H. & Nowakowski, M., (2000). The effect of arable field margin composition on invertebrate biodiversity. *Biological Conservation* **106**, 259-271.

Meffe, G.K. & Carroll, C.R., (1994). *Principles of conservation biology*. 2nd Edition. Sinauer Publishers, Massachusetts, United States of America.

Memmott, J., (1999). The structure of a plant-pollinator food web. *Ecology Letters* **2**, 276-280.

Mentis, M.T., Meiklejohn, M.J. & Scotcher, J.S.B., (1974). Veld burning in Giant's Castle Game Reserve, Natal Drakensberg. *Proceedings of the Grassland Society of South Africa* **9**, 26-31.

Mentis, M.T. & Tainton, N.M., (1984). The effect of fire on forage production and quality. In: *Ecological effects of fire in South African ecosystems* (Eds. De V. Booysen, P. & Tainton, N.M.) Springer-Verlag, Berlin, Germany. pp. 245-254.

Mielke, P.W.J., (1984). Meteorological applications of permutation techniques based on distance functions. In: *Handbook of statistics*. (Eds. Krishnaiah, P.R. & Sen, P.K.), Elsevier Science Publishers, New York, pp. 813-830.

Milchunas, D.G. & Lauenroth, W.K. (1993). Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs* **63**, 327-366.

Mills, L.S., Soulé, M.E. & Doak, D.F., (1993). The keystone-species concept in ecology and conservation. *BioScience* **43**, 219-224.

- Miller, R.H. & Onsager, J.A., (1991). Grasshopper (Orthoptera: Acrididae) and plant relationships under different grazing intensities. *Environmental Entomology* **20**, 807-814
- Mitchell, B., (1963). Ecology of two carabid beetles, *Bembidion lampros* (Herbst) and *Trechus quadristriatus* (Schrank). II. Studies on populations of adults in the field, with specific reference to the technique of pitfall trapping. *Journal of Animal Ecology* **32**, 377-392.
- Mirov, N.T., (1967). *The genus Pinus*. Ronald, New York, United States of America.
- Morris, M.G., (1981a). Responses of grassland invertebrates to management by cutting. III. Adverse effects on Auchenorrhyncha. *Journal of Applied Ecology* **18**, 107-123.
- Morris, M.G., (1990). The Hemiptera of two sown calcareous grasslands. II. Differences between treatments. *Journal of Applied Ecology* **27**, 379-393.
- Morris, M.G., (1991). The management of reserves and protected areas. In: *The scientific management of temperate communities for conservation*. (Eds. Spellerberg, I.F., Goldsmith, F.B. & Morris, M.G.). Blackwell Scientific Publications, Oxford, United Kingdom. pp. 323-347.
- Morris, M.G. & Lakhani, K.H., (1979). Responses of grassland invertebrates to management by cutting. I. Species diversity of Hemiptera. *Journal of Applied Ecology* **16**, 77-98.
- Morris, M.G. & Rispin, W.E., (1987). Abundance and diversity of the coleopterous fauna of a calcareous grassland under different cutting regimes. *Journal of Applied Ecology* **24**, 451-465.
- Morris, M.G. & Thomas, J.A., (1991). Progress in the conservation of butterflies. *Nota Lepidoptera* **2**, 32-44.

Mulkern, G.B., 1967. Food selection by grasshoppers. *Annual Review of Entomology* **12**, 59-78.

Mullen, K., Fahy, O. & Gormally, M., (2003). Ground flora and associated arthropod communities of forest road edges in Connemara, Ireland. *Biodiversity and Conservation* **12**, 87-101.

Mullie, W.C. & Kieth, J.O., (1993). The effects of aerially applied fenitrothion and chlorpyrifos on the birds in the savanna of north Senegal. *Journal of Applied Ecology* **30**, 536-550.

Muralirangan, M.C., Suresh, P. & Dhang, P.P., (1993). Observation on the grasshopper species diversity, density and distributional pattern in peninsular India. *The Entomologist (London)* 201-210.

Mushinsky, H.R. & Gibson, D.J., (1991). The influence of fire periodicity on habitat structure. In: *Habitat structure: the physical arrangement of objects in space* (Eds. Bell, S.S., McCoy, E.D. & Mushinsky, H.R.). Chapman and Hall, London, United Kingdom. pp. 237-259.

Neke, K.S. & du Plessis, M.A., (2004). The threat of transformation: quantifying the vulnerability of grasslands in South Africa. *Conservation Biology* **18**, 466-477.

Nekola, J.C., (2002). Effects of fire management on the richness and abundance of central North American grassland land snail faunas. *Animal Biodiversity and Conservation* **25**, 53-66.

New, T.R., (1984). *Insect Conservation: an Australian perspective*. Junk, Dordrecht, The Netherlands.

New, T.R., (1991). *Butterfly conservation*. Oxford University Press, Melbourne, Australia.

New, T.R. (1997). *Butterfly Conservation*. 2nd edition. Oxford University Press, Melbourne.

- New, T.R., Pyle, R.M., Thomas, J.A., Thomas, C.D. & Hammond, P.C., (1995). Butterfly conservation management. *Review of Entomology* **40**, 57-83.
- Niemelä, J., Halme, E., Pajunen, T. & Haila, Y., (1986b). Sampling spiders and carabid beetles with pitfall traps: the effect of increased sampling effort. *Annales Entomologici Fennica* **52**, 109-111.
- Niemelä, J., Halme, E. & Haila, Y., (1990). Balancing sampling effort in pitfall trapping of carabid beetles. *Entomologici Fennica* **1**, 233-238.
- Niemelä, J., Spence, J.R., Langor, D.W., Haila, Y. & Tukia, H., (1993b). Logging and boreal ground-beetle assemblages on two continents: implications for conservation. In: *Perspectives of insect conservation*. (Eds. Gaston, K.J., New, T.R. & Samways, M.J.). Intercept, Andover, United Kingdom. pp. 29-50.
- Niemelä, J. & Baur, B., (1998). Threatened species in a vanishing habitat: plants and invertebrates in calcareous grasslands in the Swiss Jura mountains. *Biodiversity and Conservation* **7**, 1407-1416.
- Noss, R.F., (1990). Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology* **4**, 355-364.
- Novacek, M.J. & Cleland, E.E., (2001). The current biodiversity extinction event: scenarios for mitigation and recovery. *Proceedings of the National Academy of Sciences of the United States of America* **98**, 5466-5470.
- Novotny, V. & Basset, Y., (2000). Rare species in communities of tropical insect herbivores: pondering the mystery of singletons. *Oikos* **89**, 564-572.
- O'Connor, T.G.O., (1996). What role for biodiversity in the sustainable use of rangelands in developing countries? In: *Rangelands in a sustainable biosphere. Proceedings of the 5th International rangeland congress*. Society of Range Management, Colorado **1**, 38-42.

O'Connor, T.J., (2001). Personal Communication.

O'Connor, T.J. & Bredenkamp, G.J., (1997). Grassland. In: *Vegetation of southern Africa*. (Eds. Cowling, R.M., Richardson, D.M. & Pierce, S.M.). Cambridge University Press, Cambridge, United Kingdom. pp. 215-257.

Ojima, D, Schimel, D.S., Parton, W.J. & Owensby, C.E., (1994). Long and short-term effects of fire on nitrogen cycling in tallgrass prairie. *Biogeochemistry* **24**, 67-84.

Oliver, I. & Beattie, A.J., (1996). Invertebrate morphospecies as surrogates for species: a case study. *Conservation Biology* **10**, 99-109.

Oliver, I., Beattie, A.J. & York, A., (1998). Spatial fidelity of plant, vertebrate and invertebrate assemblages in multiple-use forest in eastern Australia. *Conservation Biology* **12**, 822-835.

Oliver, M.D.N., Short, N.M.R. & Hanks, J., (1978). Population ecology of Oribi, Grey Rhebuck and Mountain Reedbuck in Highmoor state forest land. *South African Journal of Wildlife Research* **8**, 95-105.

Olsen D.M. & Dinerstein, E., (1998). The global 2000: a representation approach to conserving the earth's most biologically valuable ecoregions. *Conservation Biology* **12**, 502-515.

Onsager, J.A., (1963). *Characteristics of eggs and egg-pods of North Dakota Acrididae and studies of sites selected for oviposition*. PhD Thesis. North Dakota State University, North Dakota, United States of America.

Onsager, J.A., (2000). Suppression of grasshoppers in the Great Plains through grazing management. *Journal of Range Management* **53**, 592-602.

- Oostermeijer, J.G.B. & van Swaay, C.A.M., (1998). The relationship between butterflies and the environmental indicator values: a tool for conservation in a changing landscape. *Biological Conservation* **86**, 271-280.
- Opler, P.A. & Krizek, G.O., (1984). *Butterflies east of the Great Plains*. John Hopkins University Press, Baltimore, United States of America.
- Orbtel, R., (1971). Number of pitfall traps in relation to the structure of the catch of soil surface Coleoptera. *Acta Entomologica Bohemoslovaca* **68**, 300-309.
- Osem, Y., Perevolotsky, A. & Kigel, J., (2004). Site productivity and plant size explain the response of annual species to grazing exclusion in a Mediterranean semi-arid rangeland. *Journal of Ecology* **92**, 297-309.
- Otte, D., (1976). Species richness patterns of New World desert grasshoppers in relation to plant diversity. *Journal of Biogeography* **3**, 197-209.
- Otte, D. & Joern, A., (1975). Insect territoriality and its evolution – population studies of desert grasshoppers on creosote bushes. *Journal of Animal Ecology* **44**, 29-54.
- Paine, R.T., (1969). A note on trophic complexity and community stability. *The American Naturalist* **103**, 91-93.
- Panzer, R., (2002). Compatibility of prescribed burning with the conservation of insects in small isolated prairie reserves. *Conservation Biology* **16**, 1296-1307.
- Panzer, R., (2003). Importance of in situ survival, recolonization, and habitat gaps in the postfire recovery of fire-sensitive prairie insect species. *Natural Areas Journal* **23**, 14-21.
- Panzer, R. & Schwartz, M.W., (1998). Effectiveness of a vegetation-based approach to insect conservation. *Conservation Biology* **12**, 693-702.

- Panzer, R., Stillwargh, D., Gnaediger, R. & Derkovitz, G., (1995). Prevalence of remnant dependence among prairie and savanna-inhabiting insects of the Chicago region. *Natural Areas Journal* **15**, 101-116.
- Palmer, M.W. & White, P.S., (1994). Scale dependence and the species-area relationship. *American Naturalist* **144**, 717-740.
- Parmesan, C., (1996). Climate and species' range. *Nature* **382**, 765-766.
- Parmesan, C., Ryrholm, N., Stefanescu, C., Hill, J.K., Thomas, C.D., Descimon, H., Huntley, B., Kaila, L., Kullberg, J., Tammaru, T., Tennent, W.J., Thomas, J.A. & Warren, M., (1999). Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature* **399**, 579-583.
- Pearce-Kelly, P., Jones, R., Clarke, D., Walker, C., Atkin, P. & Cunningham, A.A., (1998). The captive rearing of threatened Orthoptera: a comparison of the conservation potential and practical considerations of two species' breeding programmes at the Zoological Society of London. *Journal of Insect Conservation* **2**, 201-210.
- Pearson, D.L., (1994). Selecting indicator taxa for the quantitative assessment of biodiversity. *Philosophical Transactions of the Royal Society of London Series* **345**, 75-79.
- Pearson, D.L. & Carroll, S.S., (1998). Spatial modeling of butterfly species richness using tiger beetles (Cicindelidae) as a bioindicator taxon. *Ecological Applications* **8**, 531-543.
- Petanidou, T., Vokou, D. & Margaris, N.S., (1991). *Panaxia quadripunctaria* in the highly touristic valley of butterflies (Rhodes, Greece): conservation problems and remedies. *Ambio* **20**, 124-128.
- Phillips, J.F.V., (1930). Fire: its influence on biotic communities and physical factors in South and East Africa. *South African Journal of Science* **27**, 352-367.

- Pianka, E.R., (1966). Latitudinal gradients in species diversity: a review of concepts. *The American Naturalist* **100**, 33-46.
- Picker, M., Griffiths, C. & Weaving, A., (2002). *Field guide to insects of South Africa*. Struik Publishers, Cape Town, South Africa.
- Pimentel, D., Stachow, V., Taknes, D.A., Brubaker, H.W., Dumas, A.R., Meaney, J.J., O'Neil, J.A.S., Onsi, D.E. & Corzilius, D.B., (1992). Conserving biological diversity in agricultural/forestry systems. *Bioscience* **40**, 354-362.
- Pitt, W.C., (1999). Effects of Multiple vertebrate predators on grasshopper habitat selection: trade-offs due to predation risk, foraging, and thermoregulation. *Evolutionary Ecology* **13**, 499-515.
- Polhemus, D.A., (1993). Conservation of aquatic insects. Worldwide crisis of localized threats. *American Zoologist* **33**, 588-598.
- Ponder, W.F., Colgan, D.J., Clark, G.A., Miller, A.C. & Terzis, T., (1994). Microgeographic, genetic and morphological differentiation of freshwater snails – the Hydrobiidae of Wilsons Promontory, Victoria, southeastern Australia. *Australian Journal of Zoology* **42**, 557-678.
- Pollard, E., (1977). A method for assessing changes in abundances of butterflies. *Biological Conservation* **12**, 115-131.
- Pollard, E., (1984). Synoptic studies on butterfly abundance. In: *The biology of butterflies*. (Eds. Vane-Wright, R.I. & Ackery, P.R.). Academic Press, London, United Kingdom. pp. 59-61.
- Pollard, E. & Yates, T.J., (1993). *Monitoring butterflies for ecology and conservation*. Chapman and Hall, London, United Kingdom.

Porter, E.E. & Redak, R.A., (1996). Short-term recovery of the grasshopper communities (Orthoptera: Acrididae) of a California native grassland after prescribed fire. *Environmental Entomology* **25**, 987-992.

Porter, E.E. & Redak, R.A., (1997). Diet of migratory grasshopper (Orthoptera: Acrididae) in a California native grassland and the effect of prescribed spring burning. *Environmental Entomologist* **26**, 234-240.

Pott, R.McC., (1997). Plantation forestry in South Africa and its impact on biodiversity and water. *South African Forestry Journal* **180**, 45-48.

Poulter, A.G., (1993). *A leaf key to some common grasses of the Natal Drakensberg*. South African Forestry Research Institute, South Africa.

Port, G.R. & Thompson, J.R., (1980). Outbreaks of insect herbivores on plants along motorways in the United Kingdom. *Journal of Applied Ecology* **17**, 649-656.

Power, M.E., Tilman, D. Estes, J.A., Menge, B.A., Bond, W.J., Mills, L.S., Daily, G., Castilla, J.C., Lubchenko, J. & Paine, R.T., (1996). Challenges in the quest for keystones. *BioScience* **46**, 609-620.

Prendergast, J.R., Quinn, R.M., Lawton, J.H., Eversham, B.C. & Gibbons, D.W. (1993). Rare species, the coincidence of diversity hotspots and conservation strategies. *Nature* **365**, 335-337.

Prendini, L., Theron, L-J., van der Merwe, K. & Owen-Smith, N., (1996). Abundance and guild structure of grasshoppers (Orthoptera: Acridoidea) in communally grazed and protected savanna. *South African Journal of Zoology* **31**, 120-130.

Pringle, E.L.L., Henning, G.A. & Ball, J.B., (1994). *Pennington's butterflies of southern Africa*. 2nd Edition. Struik Winchester, Cape Town, South Africa.

Pryke, S.R. & Samways, M.J., (2003). Quality of remnant grassland linkages for adult butterflies (Lepidoptera) in an afforested African landscape. *Biodiversity and Conservation* **12**, 1985-2004.

Quinn, M.A. & Walgenbach, D.D., (1990). Influence of grazing history on the community structure of grasshoppers of mixed-grass prairie. *Environmental Entomology* **19**, 1756-1766.

Quinn, M.A., Kepner, R.L., Walgenbach, D.D., Bohls, R.A., Pooler, P.D., Foster, R.N., Rueter, K.C. & Swain, J.L., (1991). Habitat characteristics and grasshopper community dynamics on mixed-grass rangeland. *The Canadian Entomologist* **123**, 89-105.

Quinn, M.A., Johnson, P.S., Butterfield, C.H. & Walgenbach, D.D., (1993). Effect of grasshopper (Orthoptera: Acrididae) density and plant composition on growth and destruction of grasses. *Environmental Entomology* **22**, 993-1002.

Raven, P.H. & Wilson, E.O., (1992). A fifty-year plan for biodiversity surveys. *Science* **258**, 1099-1100.

Rebelo, A.G., (1997). Conservation. In: *Vegetation of southern Africa*. (Eds. Cowling, R.M., Richardson, D.M. & Pierce, S.M.). Cambridge University Press, Cambridge, United Kingdom. pp. 571-589.

Reed, C.C., (1997). Responses of prairie insects and other arthropods to prescription burns. *Natural Areas Journal* **17**, 380-385.

Reid, W.V., (1998). Biodiversity hotspots. *Trends in Ecological Evolution* **13**, 275-279.

Reid, W.V. & Miller, K.R., (1989). *Keeping options alive: the scientific basis for conserving biodiversity*. World Resources Institute, Washington, United States of America.

Rentz, D.C.F., (1978). Orthoptera. In: *Biogeography and ecology of southern Africa*. (Ed. Werger, M.J.A.). W. Junk, The Hague.

Rentz, D.C.F & Weissman, D.B., (1981). Faunal affinities, systematics and bionomics of the Orthoptera of California Channel Islands. *University of California Publications, Entomology* **94**, 1-240.

Resh, V.H. & Jackson, J.K., (1993). Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. In: *Freshwater biomonitoring and benthic macroinvertebrates*. (Eds. Rosenberg, D.M. & Resh, V.M.) Chapman and Hall, New York, United States of America. pp. 195-233.

Reyers, B., Fairbanks, D.H.K. van Jaarsveld, A.S. & Thompson, M., (2001). Priority areas for the conservation of South African vegetation: a coarse filter approach. *Diversity and Distributions* **7**, 77-96.

Reyers, B., Fairbanks, D. H. K., Wessels, K. J. & van Jaarsveld, A.S., (2002a). A multicriteria approach to reserve selection: addressing long-term biodiversity maintenance. *Biodiversity and Conservation* **11**, 769-793.

Rice, L.A., (1932). Effect of fire on animal communities. *Ecology* **13**, 392-401.

Riechert, S.B. & Reeder, W.C., (1970). Effects of fire on spider distribution in southwestern Wisconsin prairies. In: *Second Northern Illinois Prairie Workshop*. (Ed. Zimmerman, J.H.). Rock Valley College, Illinois, United States of America.

Rivers-Moore, N.A. & Samways, M.J., (1996). Game and cattle trampling, and impacts of human dwellings on arthropods at a game park boundary. *Biodiversity and Conservation* **5**, 1545-1556.

Robertson, P.A., Clarke, S.A. & Warren, M.S., (1995). Woodland management and butterfly diversity. In: *Ecology and conservation of butterflies*. (Ed. Pullin, A.S.). Chapman and Hall, London, United Kingdom. pp. 113-122.

Robinson, E.T., Gibbs-Russel, G.E., Trollope, W.S.W. & Downing, B.H., (1979). Assessment of short-term burning treatments on the herb layer of False Thornveld of the eastern Cape. *Proceedings of the Grassland Society of South Africa* **14**, 79-84.

- Robinson, G.R., Holt, R.D., Gaines, M.S., Hamburg, S.P., Johnson, M.L., Fitch, H.S. & Martinko, E.A., (1992). Diverse and contrasting effects of habitat fragmentation. *Science* **257**, 524-526.
- Rodell, C.F., (1997). A grasshopper model for a grassland ecosystem. *Ecology* **58**, 227-245.
- Rohlf, R.R., & Sokal, F.J., (1995). *Biometry: the principles and practices of statistics in biological research*. 3rd Edition. Freeman and Company, New York, United States of America.
- Roux, P.W. & Opperman, D.P.J., (1986). Soil Erosion. In: *The Karoo Biome: a preliminary synthesis. Part 1- physical environment*. (Eds. Cowling, R.M., Roux, P.W. & Pieters, A.J.H.). South African National Scientific Programme Report, Pretoria. **124**, pp. 92-111.
- Rowe-Rowe, D.T. & Lowry, P.B., (1982). Influence of fire on small mammal populations in the Natal Drakensberg. *South African Journal of Wildlife Research* **12**, 130-139.
- Rozenweig, M.L., (1995). *Species diversity in space and time*. Cambridge University Press, New York, United States of America.
- Rozenweig, M.L. & Abramsky, Z., (1993). How are productivity and diversity related? Species diversity in ecological communities. In: *Historical and geographical perspectives*. (Eds. Ricklefs, E.R. & Schluter, D.). Chicago University Press, Chicago, United States of America. pp. 52-65.
- Rushton, S.P., Eyre, M.D. & Luff, M.L., (1990). The effects of scrub management on the ground beetles of oolitic limestone grassland at Castor Highlands national nature reserve, Cambridgeshire, United Kingdom. *Biological Conservation* **51**, 97-111.

Sala, O.E., Chapin, F.S. III, Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. & Wall, D.H., (2000). Global biodiversity scenarios for the year 2100. *Science* **287**, 1770-1774.

Samways, M.J., (1990). Land forms and winter habitat refugia in the conservation of montane grasshoppers in South Africa. *Conservation Biology* **4**, 375-382.

Samways, M.J., (1993). Dragonflies (Odonata) in taxic overlays and biodiversity conservation. In: *Perspectives on insect conservation*. (Eds. Gaston, K.J., New, T.R. & Samways, M.J.). Intercept, Andover, United Kingdom. pp. 111-123.

Samways, M.J., (1994). *Insect conservation biology*. Chapman and Hall, London, United Kingdom.

Samways, M. J., (1995). Southern hemisphere insects: their variety and the environmental pressures upon them. In: *Insects in a changing environment*. (Eds. Harrington, R. & Stork, N.E.). Academic Press, London, United Kingdom. pp. 297-320.

Samways, M.J., (1997). Conservation Biology of Orthoptera. In: *The bionomics of grasshoppers, katydids and their kin*. (Eds. Gangwere, S.K., Muralirangan, M.C. & Muralirangan, M.). CAB International, Wallingford, United Kingdom. pp. 481-496.

Samways, M.J., (1999b). Landscape triage for conserving insect diversity. In: *The other 99%: the conservation and biodiversity of invertebrates*. (Eds. Ponder, W. & Lunney, D.). The Royal Zoological Society of New South Wales, Mosman, Australia. pp. 269-273.

Samways, M.J., (2002a). Caring for the multitude. *Biodiversity and Conservation* **11**, 341-343.

Samways, M.J. & Kreuzinger, K., (2001). Vegetation, ungulate and grasshopper interactions inside vs. outside an African savanna game park. *Biodiversity and Conservation* **10**, 1963-1981.

Samways, M.J. & Lockwood, J.A., (1998). Orthoptera conservation: pests and paradoxes. *Journal of Insect Conservation* **2**, 143-149.

Samways, M.J. & Moore, S.D., (1991). Influence of exotic conifer patches on grasshopper (Orthoptera) assemblages in a grassland matrix at a recreational resort, Natal, South Africa. *Biological Conservation* **57**, 205-219.

Samways, M.J. & Sergeev, M.G., (1997). Orthoptera and landscape change. In: *The bionomics of grasshoppers, katydids and their kin*. (Eds. Gangwere, S.K., Muralirangan, M.C. & Muralirangan, M.). CAB International, Wallingford, United Kingdom. pp. 147-162.

Saunders, D.A., Hobbs, R.J., & Margules, C.R., (1991). Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* **5**, 18-32.

Savory, A. & Butterfield, J., (1999). *Holistic management: a new framework for decision-making*. Island Press, Washington, United States of America.

Sax, D.F. & Gaines, S.D., (2003). Species diversity: from global decreases to local increases. *Trends in Ecology and Evolution* **18**, 561-566.

Scheffer, M. & Carpenter, S.R., (2003). Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends in Ecology and Evolution* **18**, 648-656.

Scholtz, C.H. & Chown, S.L., (1993). Insect conservation and extensive agriculture: the savanna of southern Africa. In: *Perspectives on insect conservation*. (Eds. Gaston, K.J., New, T.R. & Samways, M.J.). Intercept, Andover, United Kingdom. pp. 75-95.

Scott, J.D., (1955). Principles of pasture management. In: *The grasses and pastures of South Africa*. (Ed. Meredith, D.). Central News Agency, Johannesburg, South Africa. pp. 601-623.

Scott, J.D., (1970). Pros and cons of eliminating veld fires. *Proceedings of the Grassland Society of South Africa* **5**, 23-26.

Scott, J.D., (1984). A historical review of research on fire in South Africa. In: *Ecological effects of fire in South African ecosystems*. (Eds. Booysen, P. de V. & Tainton, N.M.). Springer-Verlag, Berlin, Germany. pp. 53-65.

Scott-Shaw, R., (2002). What is Mistbelt Grassland? Notes from a talk given by Rob Scott-Shaw at KwaZulu-Natal Nature Conservation Head Office, Pietermaritzburg, South Africa.

Seastedt, T.R., Briggs, J.M. & Gibson, D.J., (1991). Controls of N limitation in tallgrass prairie. *Oecologia* **87**, 72-79.

Sergeev, M.G., (1997). Ecogeographical distribution of Orthoptera. In: *The bionomics of grasshoppers, katydids and their kin*. (Eds. Gangwere, S.K., Muralirangan, M.C. & Muralirangan, M.). CAB International, Wallingford, United Kingdom. pp. 129-146.

Sergeev, M.G., (1998). Conservation of orthopteran biological diversity relative to landscape change in temperate Eurasia. *Journal of Insect Conservation* **2**, 247-252.

Shackleton, C.M., (2000). Comparison of plant diversity in protected and communal lands in the Bushbuckridge lowveld savanna, South Africa. *Biological Conservation* **94**, 273-285.

Shapiro, A.M., (1975). The temporal component of butterfly species diversity. In: *Ecology and evolution of communities*. (Eds. Cody, M.L. & Diamond, J.M.) Belknap Press, Massachusetts, United States of America.

Shapiro, A.M., (1996). Status of butterflies. In: *Sierra Nevada ecosystem project: final report to congress. Assessments and scientific basis for management options. (Volume II)*. University of California, Centers for Water and Wildland Resources, California, United States of America.

Shreeve, T.G. & Mason, C.F., (1980). The number of butterfly species in woodlands. *Oecologia* **45**, 414-418.

Simberloff, D., (1987). The spotted owl fracas: mixing academic, applied and political ecology. *Ecology* **68**, 766-772.

Simberloff, D., (1998). Flagships, umbrellas and keystones: is single-species management passé in the landscape era? *Biological Conservation* **83**, 247-257

Simonson, S.E., Opler, P.A., Stohlgren, T.J. & Chong, G.W., (2001). Rapid assessment of butterfly diversity in a montane landscape. *Biodiversity and Conservation* **10**, 1369-1386.

Skinner, K.M., (2000). The past, present and future of rangeland grasshopper management. *Rangelands* **22**, 24-28.

Smart, S. M., Bunce, R. G. H., Firbank, L. G. & Coward, P., (2002). Do field boundaries act as refugia for grassland plant species diversity in intensively managed agricultural landscapes in Britain? *Agriculture, Ecosystems and Environment* **91**, 73-87.

Smith, K.D., (1974). The utilisation of gum trees by birds in Africa. *Ibis* **116**, 155-164.

Snyman, H.A., (2002). Fire and the dynamics of a semi-arid grassland: influence on soil characteristics. *African Journal of Range and Forage Science* **19**, 137-145.

Snyman, H.A., (2004). Estimating the short-term impact of fire on rangeland productivity in a semi-arid climate of South Africa. *Journal of Arid Environments* **59**, 685-697.

Soberón, J., Rodríguez, P. & Vasquez-Dominguez, P., (2000). Implications of the hierarchical structure of biodiversity for the development of ecological indicators of sustainable use. *Ambio* **29**, 136-142.

Solem, A. & McKenzie, N.L., (1991). The composition of land snail assemblages in the Kimberley rainforests. In: *Kimberley rainforests*. (Eds. McKenzie, N.L., Johnston, R.B. & Kendrick, P.G.) Surrey Beatty and Sons, Chipping Norton, United Kingdom. pp. 247-263.

Sotherton, N.W. & Self, M.J., (2000). Changes in plant and arthropod biodiversity on lowland farmland: an overview. In: *Ecology and Conservation of Lowland Farmland Birds*. (Eds. Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A.). Proceedings of the British Ornithologists Union Spring Conference, Tring. pp. 26-35.

Soulé, M.E. & Kohm, K.A., (1989). *Research priorities for conservation biology*. Island Press, Washington, United States of America.

Southwood, T.R.E., (1978). *Ecological methods*. Chapman and Hall, New York, United States of America.

Spellerberg, I.F., (1991). *Monitoring ecological change*. Cambridge University Press, Cambridge, United Kingdom.

Spellerberg, I.F. & Sawyer, J.W.D., (1996). Standards for biodiversity: a proposal based on biodiversity standards for forestry plantations. *Biodiversity and Conservation*. **5**, 447-459.

Spence, J.R., & Niemelä, J., (1994). Sampling carabid assemblages with pitfall traps: the madness and the method. *The Canadian Entomologist* **126**, 881-894.

Spence, J.R., Langor, D.W., Niemelä, J., Carcamo, H.A. & Currie, C.R., (1996). Northern forestry and carabids, the case for concern about old-growth species. *Annales Zoologici Fennici* **33**, 173-184.

Spitzer, K., Novotny, V., Tonner, M. & Leps, J., (1993). Habitat preferences, distribution and seasonality of the butterflies in a montane tropical rain forest, Vietnam. *Journal of Biogeography* **20**, 109-121.

Stebaev, I.V., 1970. Periodic changes in the ecological distribution of grasshoppers in the temperate and extreme continental Steppe regions, and their importance for the local ecosystems. In: *Proceedings of the international study conference on current and future problems of acridology*. Centre for Overseas Pest Research, London. pp. 207-212.

Steffan-Dewenter, I. & Tschardtke, T., (1997). Early succession of butterfly and plant communities on set-aside fields. *Oecologia* **109**, 294-302.

Stein, E.D. & Ambrose, R.F., (1998). A rapid impact assessment method for use in a regulatory context. *Wetlands* **18**, 379-392.

Stewart, D.A.B. & Brown, H.D., (1995). Locust control: assessing the environmental impact. *Plant Protection News* **40**, 3-4.

Stewart, D.A.B., (1998). Non-target grasshoppers as indicators of the side effects of chemical locust control in the Karoo, South Africa. *Journal of Insect Conservation* **2**, 263-276.

Steytler, N.S. & Samways, M.J., (1995). Biotope selection by adult male dragonflies (Odonata) at an artificial lake created for insect conservation in South Africa. *Biological Conservation* **72**, 381-386.

Stockwell, C.A., Hendry, A.P. & Kennison, M.J., (2003). Contemporary evolution meets conservation biology. *Trends in Ecology and Evolution* **18**, 94-101.

Sutton, S.L. & Collins, N.M., (1991). Insects and tropical forest conservation. In: *The conservation of insects and their habitats*. (Eds. Collins, N.M. & Thomas, J.A.). Academic Press, London, United Kingdom. pp. 405-424.

- Swengel, A.B., (1990). Monitoring butterflies using the fourth of July butterfly count. *American Midland Naturalist* **124**, 395-406.
- Swengel, A.B., (1995). Observations of spring larvae of *Lycaeides Melissa samuelis* (Lepidoptera: Lycaenidae) in central Wisconsin. *Great Lakes Entomologist* **28**, 155-170.
- Swengel, A.B., (1996a). Effects of fire and hay management on abundance of prairie butterflies. *Biological Conservation* **85**, 73-85.
- Swengel, A.B., (1996b). Observation of *Incisalia irus* (Lepidoptera: Lycaenidae) in central Wisconsin 1988-1995. *Great Lakes Entomologist* **29**, 47-62.
- Swengel, A.B., (2001). A literature review of insect responses to fire, compared to other conservation managements of open habitat. *Biodiversity and Conservation* **10**, 1149-1161.
- Sword, G.A., (2002). A role for phenotypic plasticity in the evolution of aposematism. *Proceedings of the Royal Society of London Series B Biological Sciences* **269**, 1639-1644.
- Taboda, M.A. & Lavado, R.S., (1993). Influence of cattle trampling on soil porosity under alternate dry and ponded conditions. *Soil Use Management* **9**, 139-153.
- Tainton, N.M., (1984a). Introduction to the concepts of development, production and stability of plant communities. In: *Veld and pasture management in South Africa*. (Ed. Tainton, N.M.). Shuter and Shooter, Pietermaritzburg, South Africa. pp. 1-24.
- Tainton, N.M., (1984b). The ecology of the main grazing lands of South Africa. In: *Veld and pasture management in South Africa*. (Ed. Tainton, N.M.). Shuter and Shooter, Pietermaritzburg, South Africa. pp. 25-56.
- Tainton, N.M., (1984c). Veld burning. In: *Veld and pasture management in South Africa*. (Ed. Tainton, N.M.). Shuter and Shooter, Pietermaritzburg, South Africa. pp. 363-381.

Swengel, A.B., (1996a). Effects of fire and hay management on abundance of prairie butterflies. *Biological Conservation* **85**, 73-85.

Swengel, A.B., (1996b). Observation of *Incisalia irus* (Lepidoptera: Lycaenidae) in central Wisconsin 1988-1995. *Great Lakes Entomologist* **29**, 47-62.

Swengel, A.B., (2001). A literature review of insect responses to fire, compared to other conservation managements of open habitat. *Biodiversity and Conservation* **10**, 1149-1161.

Sword, G.A., (2002). A role for phenotypic plasticity in the evolution of aposematism. *Proceedings of the Royal Society of London Series B Biological Sciences* **269**, 1639-1644.

Taboda, M.A. & Lavado, R.S., (1993). Influence of cattle trampling on soil porosity under alternate dry and ponded conditions. *Soil Use Management* **9**, 139-153.

Tainton, N.M., (1984a). Introduction to the concepts of development, production and stability of plant communities. In: *Veld and pasture management in South Africa*. (Ed. Tainton, N.M.). Shuter and Shooter, Pietermaritzburg, South Africa. pp. 1-24.

Tainton, N.M., (1984b). The ecology of the main grazing lands of South Africa. In: *Veld and pasture management in South Africa*. (Ed. Tainton, N.M.). Shuter and Shooter, Pietermaritzburg, South Africa. pp. 25-56.

Tainton, N.M., (1984c). Veld burning. In: *Veld and pasture management in South Africa*. (Ed. Tainton, N.M.). Shuter and Shooter, Pietermaritzburg, South Africa. pp. 363-381.

Tainton, N.M., Booysen, P. de V., Bransby, D.I. & Nash, R.C., (1978). Long-term effects of burning and mowing on tall grassveld in Natal: dry matter production. *Proceedings of the Grassland Society of South Africa* **13**, 41-44.

Tainton, N.M. & Camp, K.G.T., (1999). Bioresource groups 5,6 and 7: Moist and Dry Midlands Mistbelt and Northern Mistbelt. In: *Veld in KwaZulu-Natal*. (Eds. Hardy, M.B. & Hurt, C.R.). KwaZulu-Natal Department of Agriculture. pp. 81-87.

Tainton, N.M. & Mentis, M.T., (1984). Fire in grassland. In: *Ecological effects of fire in South African ecosystems*. (Eds. Booysen, P. de V. & Tainton, N.M.). Springer-Verlag, Berlin, Germany. pp. 115-147.

Tarboton, W.R., (1997a). South Africa's grasslands: the cinderella biome. *Africa – Birds & Birding* 2, 57-60.

Taylor, B., (2004). Personal Communication.

Taylor, R.J. & Doran, N., (2001). Use of terrestrial invertebrates as indicators of the ecological sustainability of forest management under the Montreal Process. *Journal of Insect Conservation* 5, 221-231.

ter Braak, C.J.F. & Šmilauer, P., (1998). Canoco reference manual and user's guide to Canoco for windows: software for canonical community ordination (Version 4). Microcomputer Power. Ithica, New York, United States of America.

Theile, H.U., (1977). Carabid Beetles in their environments. Springer-Verlag, Berlin, Germany.

Thomas, J.A., (1983). The ecology and conservation of *Lysandra bellargus* (Lepidoptera: Lycaenidae) in Britain. *Journal of Applied Ecology* 20, 59-83.

Thomas, J.A., (1991). Rare species conservation: case studies of European butterflies. In: *The scientific management of temperate communities for conservation*. (Eds. Spellerberg, I.F., Goldsmith, F.B. & Morris, M.G.). Blackwell Science, Oxford, United Kingdom. pp. 149-197.

Thomas, C.D., (1991). Habitat use and geographic ranges of butterflies from the wet lowlands of Costa Rica. *Biological Conservation* 55, 269-281.

- Thomas, C.D., (1995). Ecology and conservation of butterfly metapopulations in the fragmented British landscape. In: *Ecology and conservation of butterflies*. (Ed. Pullin, A.S.). Chapman and Hall, London, United Kingdom. pp. 46-63.
- Thomas, C.D. & Abery, J.C.G., (1995). Estimating rates of butterfly decline from distribution maps: the effect of scale. *Biological Conservation* **73**, 59-65.
- Thomas, C.D. & Mallorie, H.C., (1985). Rarity, species richness and conservation: butterflies of the Atlas Mountains in Morocco. *Biological Conservation* **35**, 95-117.
- Thomas, C.F.G. & Marshall, E.J.P., (1999). Arthropod abundance and diversity in differently vegetated margins of arable fields. *Agriculture, Ecosystems and Environment* **72**, 131-144.
- Tilman, D., (1982). *Resource competition and community structure*. Princeton University press, Princeton, United States of America.
- Topping, C.J. & Sunderland, K.D., (1992). Limitations to the use of pitfall traps in ecological studies exemplified by a study of spiders in a field of winter wheat. *Journal of Applied Ecology* **29**, 485-491.
- Trollope, W.S.W., (1984a). Fire in savanna. In: *Ecological effects of fire in South African ecosystems*. (Eds. Booysen, P. de V. & Tainton, N.M.). Springer-Verlag, Berlin. pp. 149-176.
- Turner, M.G., (1989). Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* **20**, 171-197.
- Turner, M.G. & Gardner, R.H., (1991). *Quantitative methods in landscape ecology*. Springer-Verlag, New York, United States of America.
- Tyson, P.D. & Crimp, S.J., (1998). The climate of the Kalahari transect. *Transactions of the Royal Society of South Africa* **53**, 93-112.

- Uvarov, B.P., (1977). *Grasshoppers and locusts*, Vol. 2. Centre for Overseas Pest Research, London, United Kingdom.
- van Jaarsveld, A.S., Freitag, S., Chown, S. L., Muller, C., Koch, S., Hull, H., Bellamy, C., Krüger, M., Endrödy-Younga, S., Mansell, M. W. & Scholtz, C. H., (1998). Biodiversity assessment and conservation strategies. *Science* **279**, 2106-2108.
- van Oudtshoorn, F.P., (1992). *Guide to grasses of South Africa*. Briza Publications, Arcadia, South Africa.
- Vanderwoude, C., Andersen, A.N. & House, A.P.N., (1997). Community organization, biogeography and seasonality of ants in an open forest of south-eastern Queensland. *Australian Journal of Zoology* **45**, 523-537.
- Van Wieren, S.E., (1989). The management of populations of large mammals. In: *The scientific management of temperate communities for conservation*. (Eds. Spellerberg, I.F., Goldsmith, F.B. & Morris, M.G.). Blackwell Scientific Publications, Oxford, United Kingdom. pp. 103-127.
- van Wingerden, W.K.R.E., Musters, J.C.M. & Maaskamp, F.I.M., (1991a). The influence of cattle grazing intensity on grasshopper abundance (Orthoptera: Acrididae). *Proceedings of Experimental and Applied Entomology* **2**, 28-34.
- van Wingerden, W.K.R.E., Musters, J.C.M. & Maaskamp, F.I.M., (1991b). The influence of temperature on the duration of egg development in West European Grasshoppers (Orthoptera: Acrididae). *Oecologia* **87**, 417-423.
- van Wingerden, W.K.R.E., Musters, J.C.M., Kleukers, R.M.J.C., Bongers, W. & van Biezen, J.B., (1991). The influence of cattle grazing intensity on grasshopper abundance (Orthoptera: Acrididae). *Proceedings of Experimental and Applied Entomology* **2**, 28-34.

- van Wingerden, W.K.R.E., van Kreveld, A.R. & Bongers, W., (1992). Analysis of species composition and abundance of grasshoppers (Orthoptera: Acrididae) in natural and fertilized grasslands. *Journal of Applied Entomology*. **113**, 138-152.
- Veldtman, R., McGeoch, M.A. & Scholtz, C.H., (2002). Variability in cocoon size in southern African wild silk moths: implications for sustainable harvesting. *African Entomology*. **10**, 127-136.
- Verdu, J.R., Crespo, M.B. & Galante, E., (2000). Conservation strategy of a nature reserve in Mediterranean ecosystems: the effects of protection from grazing on biodiversity. *Biodiversity and Conservation* **9**, 1707-1721.
- Vitt, L.J., Avilapires, T.C.S., Esposito, M.C. & Miles, D.B., (2000). Niche segregation among sympatric Amazonian Teiid lizards. *Oecologia* **122**, 410-420.
- Wagge, B.E., (1985). Trapping efficiency of carabid beetles in glass and plastic pitfall traps containing different solutions. *Fauna Norvegicum Series (B)* **32**, 33-36.
- Walker, B.H., (1992). Biological diversity and ecological redundancy. *Conservation Biology* **6**, 18-23.
- Walpole, M.J. & Leader-Williams, N., (2002). Tourism and flagship species in conservation. *Biodiversity and Conservation* **11**, 543-547.
- Warren, M.S., Pollard, E. & Bibby, T.J., (1986). Annual and long term changes in a population of the wood white butterfly *Leptidea sinapis*. *Journal of Animal Ecology* **55**, 707-719.
- Warren, S.D., Scifres, C.J. & Teel, P.D., (1987). Response of grassland arthropods to burning: a review. *Agriculture, Ecosystems and Environment* **19**, 105-130.
- Webb, M.R. & Pullin, A.S., (1998). Effects of submergence by winter floods on diapausing caterpillars of a wetland butterfly, *Lycaena dispar batavus*. *Ecological Entomology* **23**, 96-99.

- Weibull, A.C., Östman, Ö. & Granqvist, Å., (2003). Species richness in agroecosystems: the effect of landscape, habitat and farm management. *Biodiversity and Conservation* **12**, 1335-1355.
- Weiss, S.B., White, R.R., Murphy, D.D. & Erlich, P.R., (1987). Growth and dispersal of larvae of the checkerspot butterfly *Euphydryas editha*. *Oikos* **50**, 161-166.
- Wettstien, W. & Schmid, B., (1999). Conservation of arthropod diversity in montane wetlands: effect of altitude, habitat quality and habitat fragmentation on butterflies and grasshoppers. *Journal of Applied Ecology* **36**, 363-373.
- Whitford, W.G., Van Zee, J., Nash, M.S., Smith, W.E. & Herrick, J.E., (1999). Ants as indicators of exposure to indicators of environmental stressors in North American desert grasslands. *Environmental Monitoring and Assessment* **54**, 143-171.
- Wiens, J.A., (1989). Spatial scaling in ecology. *Functional Ecology* **3**, 385-397.
- Wiens, J.A., (1997). The emerging role of patchiness in conservation biology. In: *The ecological basis for conservation: heterogeneity, ecosystems and biodiversity* (Eds. Pickett, S.T.A., Ostfeld, R.S., Shachak, M. & Likens, G.E.). Chapman and Hall, New York United States of America. pp. 93-107.
- Wiens, J.A., Schooley, R.L. & Weeks, R.D., (1997). Patchy landscapes and animal movements: do beetles percolate? *Oikos* **78**, 257-264.
- Wilcove, D., (1993). Getting ahead of the extinction curve. *Ecological Applications* **3**, 218-220.
- Williams, P.H. & Gaston, K.J., (1994). Measuring more of biodiversity: can higher taxon richness predict wholesale species richness? *Biological Conservation* **67**, 211-217.
- Willott, S.J. & Hassall, M., (1998). Life-history responses of British grasshoppers (Orthoptera: Acrididae) to temperature change. *Functional Ecology* **12**, 232-241.

With, K.A., (1994). Using fractal analysis to see how species perceive landscape structure. *Landscape Ecology* **9**, 25-36.

Woiwood, I.P. & Thomas, J.A., (1993). The ecology of butterflies and moths at the landscape scale. In: *Landscape ecology in Britain*. (Eds. Hannes-Young, R. & Bunce, R.G.H.). University of Nottingham Press, Nottingham, United Kingdom. pp. 76-92.

Wood, B. & Gillman, M.P., (1998). The effects of disturbance on forest butterflies using two methods of sampling in Trinidad. *Biodiversity and Conservation* **7**, 597-616.

Wood, P. A. & Samways, M. J., (1991). Landscape element pattern and continuity of butterfly flight paths in an ecologically landscaped botanic garden, Natal, South Africa. *Biological Conservation* **58**, 149-166.

Wright, D.E., Bailey, P. & Swincer, D., (1984). Pest control: recent advances and future prospects. (Eds. Bailey, P. & Swincer, D.). *Proceedings of the Fourth Australian Applied Entomological Research Conference, Adelaide*. 56-60.

Wright, M.G., (1993). Insect conservation in the African Cape Fynbos, with special reference to endophagous insects. In: *Perspectives of insect conservation*. (Eds. Gaston, K.J., New, T.R. & Samways, M.J.). Intercept, Andover, United Kingdom. pp. 97-110.

Wright, M.G. & Samways, M.J., (1998). Insect species richness tracking plant species richness in a diverse flora: gall insects in the Cape Floristic Region, South Africa. *Oecologia* **115**, 427-433.

Wright, M.G. & Samways, M.J., (1999). Plant characteristics determine insect borer assemblages on *Protea* species in the Cape Fynbos, and importance for conservation management. *Biodiversity and Conservation* **8**, 1089-1100.

Yen, A.L. & Butcher, R.J., (1997). *An overview of conservation of non-marine invertebrates in Australia*. Endangered Species Program. Environment Australia, Canberra, Australia.

York, A., (1999). Long-term effects of frequent low-intensity burning on the abundance of litter-dwelling invertebrates in coastal blackbutt forests of southeastern Australia. *Journal of Insect Conservation* **3**, 191-199.

Yoshino, M.M., 1965. *Climate in a small area*. University of Tokyo Press, Tokyo, Japan.

Zaloumis, G., (2004). Personal Communication.

Zar, J.H., (1984). *Biostatistical analysis*. 2nd Edition. Prentice-Hall, Englewood Cliffs, United States of America.

Zschokke, S., Dolt, D., Rusterholz, H-P., Oggier, P., Braschler, B., Thommen, G.H., Ludin, E., Erhardt, A. & Baur, B., (2000). Short-term responses of plants and invertebrates to experimental small-scale grassland fragmentation. *Oecologia* **125**, 559-572.

Zulka, K.P., Milasowszky, N. & Lethmayer, C., (1997). Spider biodiversity potential of an ungrazed and a grazed inland salt meadow in the National Park 'Neusiedler See-Seewinkel' (Austria): implications for management (Arachnida: Araneae). *Biodiversity and Conservation* **6**, 75-88.

APPENDIX 1. Table 1. The grasshopper species response to aspect for the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	LOW	HIGH	P- value
ACOFER	3	0	0.51
ACOTIB	18	0	0.00*
ACRACC	20	9	0.31
ACRBIC	12	1	0.10
ACRSPP	2	0	1.00
ANADRE	7	0	0.11
ANEMON	6	0	0.24
ANTGRA	4	13	0.36
CALNAT	2	1	1.00
CALSEM	19	0	0.01*
CANGRA	2	2	0.93
CATMEL	11	3	0.25
CONSP	19	23	0.74
CORSTE	14	16	0.84
CRUCRU	54	19	0.01*
CYRAER	1	10	0.24
DICSPU	5	0	0.37
DIRABB	1	15	0.06
EREBAS	19	1	0.02*
EUPCYL	0	2	1.00
EYPPLO	2	22	0.01*
FAUMIL	8	2	0.46
FAUROS	1	2	1.00
GASCRA	20	6	0.21
GASDRA	27	1	0.00*
GASVIT	3	2	0.96
GRYLL	6	0	0.24
GYMTEM	5	6	0.86

SPP. CODE	LOW	HIGH	P- value
HETGUT	0	17	0.00*
HETHER	1	2	1.00
LENMIN	1	26	0.01*
LENOBT	7	25	0.07
MACBIL	27	14	0.24
MAURUB	2	0	1.00
OCHCAF	25	0	0.00*
ORNCYA	2	2	1.00
ORTDAS	42	17	0.17
ORTROS	13	5	0.36
ORTZUL	1	1	1.00
OXYHYL	0	8	0.06
PARTRI	0	3	0.51
PARXAN	5	8	0.67
PHASUL	10	7	0.71
PHLHUM	13	5	0.32
PHLSPP	0	2	1.00
PHYLEP	2	0	1.00
PNOSQU	3	1	0.63
QACFAS	6	13	0.51
RHACEP	1	4	0.57
SCIROS	5	0	0.27
TETARC	3	2	0.96
TETTIG	2	0	1.00
TRADRA	3	0	0.50
VITBOT	17	17	0.98
WHIALT	0	9	0.09

APPENDIX 1. Table 2. The grasshopper species response to the forbaceous plant content of the 10 study sites combined (ISA value = P-value). *= Significant at $p=0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	WARM	COOL	P- value
ACOFER	0	6	0.06
ACOTIB	0	23	0.00*
ACRACC	10	20	0.37
ACRBIC	11	1	0.19
ACRSPP	1	0	1.00
ANADRE	6	0	0.21
ANEMON	0	11	0.01*
ANTGRA	3	18	0.09
CALNAT	1	1	1.00
CALSEM	14	1	0.14
CANGRA	2	3	0.75
CATMEL	7	5	0.81
CONSPP	18	23	0.68
CORSTE	14	16	0.84
CRUCRU	46	24	0.16
CYRAER	1	13	0.05*
DICSPU	6	0	0.23
DIRABB	13	1	0.17
EREBAS	14	3	0.19
EUPCYL	0	2	0.39
EYPPLO	14	5	0.33
FAUMIL	5	6	0.80
FAUROS	0	6	0.14
GASCRA	15	11	0.67
GASDRA	19	3	0.14
GASVIT	2	3	0.93
GRYLL	5	0	0.40
GYMTEM	0	19	0.00*

SPP. CODE	WARM	COOL	P- value
HETGUT	8	4	0.67
HETHER	0	6	0.14
LENMIN	2	25	0.03*
LENOBT	6	29	0.02*
MACBIL	20	21	0.83
MAURUB	0	2	0.38
OCHCAF	3	14	0.13
ORNCYA	2	3	0.78
ORTDAS	44	14	0.12
ORTROS	15	3	0.18
ORTZUL	4	0	0.26
OXYHYL	0	11	0.01*
PARTRI	0	4	0.37
PARXAN	4	8	0.63
PHASUL	4	16	0.12
PHLHUM	4	15	0.18
PHLSPP	0	2	0.41
PHYLEP	1	0	1.00
PNOSQU	0	6	0.11
QACFAS	26	0	0.00*
RHACEP	1	3	0.64
SCIROC	0	3	0.54
TETARC	2	3	0.83
TETTIG	1	0	1.00
TRADRA	3	0	0.51
VITBOT	14	22	0.40
WHIALT	6	1	0.36

APPENDIX 1. Table 3. The grasshopper species response to vegetation height of the 10 study sites combined (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	SHORT	TALL	P- value
ACOFER	2	1	0.85
ACOTIB	17	0	0.04*
ACRACC	21	8	0.26
ACRBIC	12	0	0.13
ACRSPP	1	0	1.00
ANADRE	5	0	0.42
ANEMON	6	0	0.26
ANTGRA	11	5	0.67
CALNAT	1	2	0.85
CALSEM	16	0	0.06
CANGRA	1	5	0.33
CATMEL	7	6	0.86
CONSPP	25	16	0.48
CORSTE	12	19	0.42
CRUCRU	70	5	0.00*
CYRAER	1	12	0.09
DICSPU	3	1	0.68
DIRABB	7	5	0.82
EREBAS	13	2	0.27
EUPCYL	0	3	0.31
EYPPLO	6	16	0.23
FAUMIL	10	1	0.32
FAUROS	2	1	0.89
GASCRA	21	6	0.22
GASDRA	25	0	0.02*
GASVIT	1	4	0.60
GRYLL	6	0	0.28
GYMTEM	10	1	0.27

SPP. CODE	SHORT	TALL	P- value
HETGUT	0	31	0.00*
HETHER	0	5	0.44
LENMIN	5	23	0.06
LENOBT	12	20	0.36
MACBIL	31	9	0.08
MAURUB	1	0	1.00
OCHCAF	19	0	0.02*
ORNCYA	1	5	0.58
ORTDAS	60	3	0.00*
ORTROS	23	0	0.01*
ORTZUL	2	0	0.87
OXYHYL	6	0	0.26
PARTRI	4	0	0.50
PARXAN	6	7	0.93
PHASUL	6	14	0.24
PHLHUM	13	3	0.33
PHLSPP	1	0	1.00
PHYLEP	1	0	1.00
PNOSQU	5	0	0.38
QACFAS	14	5	0.42
RHACEP	1	4	0.45
SCIROS	4	0	0.56
TETARC	2	2	1.00
TETTIG	1	0	1.00
TRADRA	2	0	0.61
VITBOT	21	12	0.49
WHIALT	2	6	0.36

APPENDIX 1. Table 4. The grasshopper species response to veld condition scores for the 10 study sites (ISA value = P-value).
 *= Significant at $p=0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	LOW	HIGH	P- value
ACOFER	0	2	0.70
ACOTIB	1	15	0.05
ACRACC	11	18	0.50
ACRBIC	1	11	0.17
ACRSPP	0	1	1.00
ANADRE	0	6	0.24
ANEMON	0	5	0.45
ANTGRA	18	2	0.10
CALNAT	1	1	1.00
CALSEM	0	17	0.03
CANGRA	3	1	0.61
CATMEL	4	8	0.62
CONSPP	21	21	0.93
CORSTE	19	12	0.43
CRUCRU	12	61	0.00
CYRAER	13	1	0.04
DICSPU	0	4	0.49
DIRABB	4	7	0.80
EREBAS	2	15	0.13
EUPCYL	2	0	0.38
EYPPLO	16	5	0.14
FAUMIL	4	6	0.78
FAUROS	3	1	0.74
GASCRA	7	19	0.28
GASVIT	3	2	0.95
GASDRA	1	25	0.01
GRYLL	0	5	0.37
GYMTEM	9	3	0.35

SPP. CODE	LOW	HIGH	P- value
HETGUT	22	0	0.00
HETHER	3	1	0.71
LENMIN	30	1	0.00
LENOBT	29	6	0.02
MACBIL	14	27	0.23
MAURUB	0	1	1.00
OCHCAF	0	22	0.00
ORNCYA	3	2	0.78
ORTROS	0	24	0.00
ORTZUL	0	3	0.62
ORTDAS	11	48	0.05
OXYHYL	11	0	0.01
PARTRI	4	0	0.39
PARXAN	10	4	0.43
PHASUL	8	9	0.99
PHLHUM	8	9	0.89
PHLSPP	2	0	0.39
PHYLEP	0	1	1.00
PNOSQU	1	2	0.92
QACFAS	3	16	0.22
RHACEP	5	1	0.34
SCIROC	0	4	0.28
TETARC	3	2	0.84
TETTIG	0	1	1.00
TRADRA	0	3	0.52
VITBOT	20	15	0.65
WHIALT	4	2	0.80

APPENDIX 1. Table 5. The grasshopper species response to Highland Sourveld versus Midlands Mistbelt grassland (ISA value = P-value). *= Significant at $p=0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	HS	MM	P- value
ACOFER	5	0	0.25
ACOTIB	18	0	0.01*
ACRACC	26	6	0.04*
ACRBIC	5	6	0.82
ACRSPP	2	0	1.00
ANADRE	7	0	0.13
ANEMON	6	0	0.23
ANTGRA	1	23	0.01*
CALNAT	3	0	0.49
CALSEM	16	1	0.08
CANGRA	2	2	0.89
CATMEL	16	1	0.03*
CONSP	6	37	0.02*
CORSTE	14	15	0.91
CRUCRU	47	25	0.11
CYRAER	1	10	0.13
DICSPU	5	0	0.38
DIRABB	1	12	0.22
EREBAS	20	1	0.01*
EUPCYL	0	2	0.47
EYPPLO	2	22	0.01*
FAUMIL	1	16	0.03*
FAUROS	0	4	0.30
GASCRA	10	16	0.54
GASDRA	17	5	0.22
GASVIT	1	4	0.59
GRYLL	2	2	1.00
GYMTEM	6	5	0.94

SPP. CODE	HS	MM	P- value
HETGUT	3	10	0.24
HETHER	2	1	1.00
LENMIN	2	24	0.03*
LENOBT	9	23	0.12
MACBIL	23	18	0.62
MAURUB	2	0	1.00
OCHCAF	23	0	0.00*
ORNCYA	1	6	0.24
ORTDAS	9	52	0.01*
ORTROS	4	13	0.39
ORTZUL	1	1	0.86
OXYHYL	0	9	0.03*
PARTRI	0	3	0.36
PARXAN	3	12	0.23
PHASUL	9	8	0.88
PHLHUM	9	8	0.89
PHLSPP	0	2	0.49
PHYLEP	2	0	1.00
PNOSQU	2	1	1.00
QACFAS	4	18	0.10
RHACEP	0	9	0.03*
SCIROS	5	0	0.23
TETARC	3	2	0.96
TETTIG	2	0	1.00
TRADRA	3	0	0.52
VITBOT	10	26	0.12
WHIALT	0	12	0.00*

APPENDIX 1. Table 6. The grasshopper species response to grazing for the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	GRAZED	NOT	P- value
ACOFER	4	0	0.28
ACOTIB	19	0	0.00*
ACRACC	36	1	0.00*
ACRBIC	4	8	0.50
ACRSPP	1	0	1.00
ANADRE	4	0	0.47
ANEMON	7	0	0.15
ANTGRA	9	10	0.88
CALNAT	2	1	0.71
CALSEM	6	8	0.83
CANGRA	7	0	0.16
CATMEL	2	16	0.04*
CONSPP	11	34	0.05*
CORSTE	19	10	0.39
CRUCRU	47	25	0.12
CYRAER	2	9	0.31
DICSPU	0	6	0.09
DIRABB	2	9	0.51
EREBAS	4	14	0.18
EUPCYL	0	2	0.38
EYPPLO	11	9	0.81
FAUMIL	2	10	0.30
FAUROS	2	2	0.92
GASCRA	12	15	0.74
GASDRA	13	7	0.59
GASVIT	1	5	0.34
GRYLL	0	8	0.04*
GYMTEM	16	0	0.02*

SPP. CODE	GRAZED	NOT	P- value
HETGUT	11	1	0.25
HETHER	2	2	0.89
LENMIN	8	15	0.43
LENOBT	14	17	0.76
MACBIL	25	15	0.42
MAURUB	1	0	1.00
OCHCAF	18	0	0.02*
ORNCYA	0	13	0.00*
ORTDAS	13	45	0.10
ORTROS	3	17	0.12
ORTZUL	2	1	0.85
OXYHYL	7	0	0.15
PARTRI	4	0	0.30
PARXAN	12	2	0.23
PHASUL	4	17	0.06
PHLHUM	9	8	0.85
PHLSPP	1	0	1.00
PHYLEP	0	2	0.39
PNOSQU	4	0	0.49
QACFAS	2	25	0.01*
RHACEP	0	6	0.20
SCIROS	4	0	0.28
TETARC	2	3	0.83
TETTIG	0	2	0.40
TRADRA	0	4	0.15
VITBOT	18	17	0.90
WHIALT	0	8	0.08

APPENDIX 1. Table 7. The grasshopper species response to burn frequency of the 10 study sites (ISA value = P-value). *= Significant at $p=0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	BIENNIAL	OTHER	P- value
ACOFER	3	0	0.76
ACOTIB	15	0	0.09
ACRACC	22	6	0.25
ACRBIC	11	1	0.31
ACRSPP	1	0	1.00
ANADRE	4	0	0.53
ANEMON	5	0	0.46
ANTGRA	19	0	0.10
CALNAT	3	0	0.62
CALSEM	16	0	0.13
CANGRA	0	7	0.08
CATMEL	16	0	0.08
CONSPP	23	20	0.80
CORSTE	17	13	0.74
CRUCRU	39	28	0.74
CYRAER	12	0	0.18
DICSPU	2	1	1.00
DIRABB	0	24	0.01*
EREBAS	15	1	0.19
EUPCYL	1	0	1.00
EYPPLO	2	29	0.01*
FAUMIL	11	1	0.33
FAUROS	4	0	0.60
GASCRA	20	6	0.37
GASDRA	20	1	0.15
GASVIT	6	0	0.40
GRYLL	5	0	0.53
GYMTEM	12	0	0.24

SPP. CODE	BIENNIAL	OTHER	P- value
HETGUT	1	19	0.02*
HETHER	4	0	0.60
LENMIN	16	7	0.55
LENOBT	23	8	0.26
MACBIL	19	24	0.57
MAURUB	1	0	1.00
OCHCAF	16	0	0.09
ORNCYA	6	0	0.43
ORTDAS	28	31	0.88
ORTROS	5	12	0.54
ORTZUL	0	6	0.10
OXYHYL	5	0	0.47
PARTRI	3	0	0.71
PARXAN	3	17	0.06
PHASUL	14	2	0.31
PHLHUM	21	0	0.05*
PHLSPP	1	0	1.00
PHYLEP	1	0	1.00
PNOSQU	4	0	0.58
QACFAS	2	35	0.00*
RHACEP	4	1	0.75
SCIROC	3	0	0.63
TETARC	6	0	0.41
TETTIG	1	0	1.00
TRADRA	2	0	1.00
VITBOT	21	11	0.54
WHIALT	0	18	0.00*

APPENDIX 1. Table 8. The grasshopper species response to disturbance at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	LOW	HIGH	P- value
ACOFER	1	2	1.00
ACOTIB	8	3	0.70
ACRACC	7	35	0.01*
ACRBIC	11	1	0.33
ACRSPP	0	4	0.19
ANADRE	0	4	0.64
ANEMON	5	0	0.46
ANTGRA	14	4	0.43
CALNAT	0	8	0.04*
CALSEM	2	18	0.03*
CANGRA	5	0	0.52
CATMEL	6	6	0.94
CONSPP	37	5	0.07
CORSTE	9	23	0.24
CRUCRU	24	49	0.13
CYRAER	7	2	0.63
DICSPU	4	0	0.57
DIRABB	11	1	0.32
EREBAS	12	2	0.45
EUPCYL	1	0	1.00
EYPPLO	16	3	0.30
FAUMIL	14	0	0.13
FAUROS	4	0	0.57
GASCRA	12	10	0.99
GASDRA	4	23	0.06
GASVIT	1	7	0.09
GRYLL	5	0	0.50
GYMTEM	7	4	0.72

SPP. CODE	LOW	HIGH	P- value
HETGUT	5	9	0.60
HETHER	1	3	0.71
LENMIN	18	3	0.35
LENOBT	17	9	0.82
MACBIL	29	11	0.21
MAURUB	1	0	1.00
OCHCAF	5	13	0.23
ORNCYA	6	0	0.43
ORTDAS	54	4	0.02*
ORTROS	13	3	0.44
ORTZUL	3	0	0.73
OXYHYL	5	0	0.49
PARTRI	2	1	1.00
PARXAN	16	0	0.09
PHASUL	9	6	0.95
PHLHUM	8	10	0.72
PHLSPP	1	0	1.00
PHYLEP	1	0	1.00
PNOSQU	4	0	0.55
QACFAS	23	0	0.03*
RHACEP	5	0	0.47
SCIROS	3	0	0.63
TETARC	6	0	0.43
TETTIG	1	0	1.00
TRADRA	2	0	1.00
VITBOT	18	17	0.93
WHIALT	7	0	0.32

APPENDIX 1. Table 9. The grasshopper species response to the season of burn at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 7 for grasshopper species codes.

SPP. CODE	SPRING	WINTER	P- value
ACOFER	4	0	0.72
ACOTIB	17	0	0.10
ACRACC	31	1	0.03*
ACRBIC	3	15	0.07
ACRSPP	1	0	1.00
ANADRE	5	0	0.51
ANEMON	6	0	0.47
ANTGRA	5	25	0.03*
CALNAT	1	2	1.00
CALSEM	9	5	0.68
CANGRA	6	0	0.47
CATMEL	10	4	0.58
CONSPP	7	37	0.07
CORSTE	14	17	0.78
CRUCRU	49	22	0.21
CYRAER	1	21	0.02*
DICSPU	5	0	0.52
DIRABB	13	0	0.16
EREBAS	14	1	0.39
EUPCYL	0	5	0.20
EYPPLO	13	5	0.46
FAUMIL	1	24	0.01*
FAUROS	1	6	0.19
GASCRA	7	28	0.05*
GASDRA	9	10	0.95
GASVIT	0	12	0.03*
GRYLL	1	3	0.67
GYMTEM	12	1	0.26

SPP. CODE	SPRING	WINTER	P- value
HETGUT	11	2	0.45
HETHER	1	6	0.21
LENMIN	5	21	0.25
LENOBT	14	21	0.40
MÁCBIL	30	8	0.22
MAURUB	1	0	1.00
OCHCAF	15	1	0.19
ORNCYA	0	17	0.01*
ORTDAS	9	49	0.09
ORTROS	3	18	0.08
ORTZUL	2	0	0.84
OXYHYL	6	0	0.46
PARTRI	4	0	0.74
PARXAN	10	2	0.47
PHASUL	4	26	0.01*
PHLHUM	7	14	0.44
PHLSPP	1	0	1.00
PHYLEP	1	0	1.00
PNOSQU	3	0	0.88
QACFAS	9	8	0.87
RHACEP	0	13	0.02*
SCIROC	4	0	0.63
TETARC	1	4	0.93
TETTIG	1	0	1.00
TRADRA	2	0	1.00
VITBOT	13	29	0.11
WHIALT	0	8	0.07

APPENDIX 2. Table 1. The butterfly species response to aspect for the 10 study sites (ISA value = P-value). *= Significant at $p=0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	WARM	COOL	P- value
ACRHOR	1	2	1.00
ACTLUC	8	6	0.87
ALOORE	6	2	0.53
ALOTAI	2	2	1.00
BELAUR	15	8	0.59
BELZOC	14	2	0.19
CASCAS	6	27	0.09
CATCLO	6	4	0.90
CATFLO	14	2	0.21
COLELE	18	10	0.46
CUPCIS	1	9	0.19
DANCHR	18	26	0.50
DINBOW	6	0	0.27
EURBRI	6	11	0.64
HARNOQ	0	9	0.05*
HYAEPO	2	0	1.00
ORAARI	0	3	0.39
ORALAC	10	0	0.15
ORASUB	0	3	0.39
PAPDEM	5	2	0.74
PAPNIR	0	6	0.17
PARPUN	2	6	0.51
PREHIE	4	0	0.51
PREOCT	12	21	0.41
PSEVAR	16	2	0.21
STYWIC	5	3	0.79
VANCAR	15	4	0.53

APPENDIX 2. Table 2. The butterfly species response to forbaceous plant content at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	LOW	HIGH	P- value
ACRHOR	4	0	0.51
ACTLUC	4	10	0.55
ALOORE	6	2	0.56
ALOTAI	6	0	0.43
BELAUR	15	7	0.55
BELZOC	3	14	0.18
CASCAS	21	9	0.39
CATCLO	5	5	1.00
CATFLO	6	10	0.69
COLELE	4	30	0.02*
CUPCIS	7	1	0.35
DANCHR	21	21	1.00
DINBOW	0	5	0.35
EURBRI	9	7	0.86
HARNOQ	2	1	1.00
HYAEPO	2	0	1.00
ORAARI	2	0	1.00
ORALAC	9	1	0.31
ORASUB	2	0	1.00
PAPDEM	13	0	0.03*
PAPNIR	1	1	1.00
PARPUN	5	2	0.64
PREHIE	4	0	0.51
PREOCT	21	12	0.47
PSEVAR	14	4	0.39
STYWIC	1	8	0.27
VANCAR	2	20	0.17

APPENDIX 2. Table 3. The butterfly species response to vegetation height at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	SHORT	TALL	P- value
ACRHOR	1	2	1.00
ACTLUC	7	7	0.97
ALOORE	11	0	0.09
ALOTAI	4	1	0.74
BELAUR	15	8	0.57
BELZOC	19	0	0.05*
CASCAS	9	24	0.21
CATCLO	9	2	0.37
CATFLO	10	5	0.61
COLELE	26	5	0.11
CUPCIS	2	7	0.43
DANCHR	23	20	0.81
DINBOW	6	0	0.40
EURBRI	7	10	0.73
HARNOQ	3	1	0.68
HYAEPO	2	0	1.00
ORAARI	0	3	0.37
ORALAC	2	8	0.41
ORASUB	0	3	0.35
PAPDEM	5	2	0.70
PAPNIR	1	2	1.00
PARPUN	1	10	0.15
PREHIE	1	2	1.00
PREOCT	16	17	0.88
PSEVAR	4	16	0.18
STYWIC	7	1	0.51
VANCAR	14	5	0.62

APPENDIX 2. Table 4. The butterfly species response to veld condition at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	LOW	HIGH	P- value
ACRHOR	1	1	1.00
ACTLUC	9	4	0.72
ALOORE	10	0	0.13
ALOTAI	1	4	0.55
BELAUR	15	8	0.49
BELZOC	19	1	0.04*
CASCAS	6	26	0.12
CATCLO	9	2	0.41
CATFLO	11	5	0.46
COLELE	27	5	0.09
CUPCIS	2	5	0.80
DANCHR	22	20	0.88
DINBOW	7	0	0.27
EURBRI	8	8	0.98
HARNOQ	1	3	0.74
HYAEPO	2	0	1.00
ORAARI	0	3	0.43
ORALAC	2	8	0.36
ORASUB	0	3	0.45
PAPDEM	4	3	0.90
PAPNIR	1	1	1.00
PARPUN	1	7	0.38
PREHIE	1	1	1.00
PREOCT	15	17	0.88
PSEVAR	6	11	0.61
STYWIC	9	0	0.21
VANCAR	18	3	0.36

APPENDIX 2. Table 5. The butterfly species response to Highland Sourveld versus Midlands Mistbelt grassland (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	HS	MM	P- value
ACRHOR	0	4	0.51
ACTLUC	1	16	0.17
ALOORE	2	6	0.67
ALOTAI	0	5	0.41
BELAUR	4	20	0.19
BELZOC	10	5	0.58
CASCAS	15	14	0.95
CATCLO	4	7	0.70
CATFLO	12	5	0.39
COLELE	12	16	0.74
CUPCIS	4	3	0.90
DANCHR	21	22	0.91
DINBOW	5	0	0.40
EURBRI	5	13	0.43
HARNOQ	2	2	1.00
HYAEPO	0	2	1.00
ORAARI	0	2	1.00
ORALAC	0	11	0.15
ORASUB	3	0	0.42
PAPDEM	0	9	0.32
PAPNIR	6	0	0.19
PARPUN	0	13	0.06
PREHIE	0	4	0.49
PREOCT	17	15	0.90
PSEVAR	7	9	0.89
STYWIC	9	1	0.19
VANCAR	21	2	0.12

APPENDIX 2. Table 6. The butterfly species response to grazing at the 10 study sites (ISA value = P-value). *= Significant at $p=0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	GRAZED	NOT GRAZED	P- value
ACRHOR	0	6	0.16
ACTLUC	1	20	0.02*
ALOORE	2	6	0.51
ALOTAI	5	0	0.41
BELAUR	7	17	0.32
BELZOC	4	13	0.24
CASCAS	23	6	0.26
CATCLO	3	9	0.42
CATFLO	14	3	0.21
COLELE	7	27	0.06
CUPCIS	2	6	0.60
DANCHR	25	17	0.55
DINBOW	0	9	0.08
EURBRI	3	17	0.11
HARNOQ	3	1	0.71
HYAEPO	0	3	0.43
ORAARI	0	3	0.43
ORALAC	8	1	0.47
ORASUB	2	0	1.00
PAPDEM	2	6	0.46
PAPNIR	4	0	0.54
PARPUN	0	14	0.03*
PREHIE	1	2	1.00
PREOCT	23	10	0.30
PSEVAR	10	7	0.77
STYWIC	5	2	0.67
VANCAR	16	4	0.45

APPENDIX 2. Table 7. The butterfly species response to biennial burning at the 10 study sites (ISA value = P-value). *= Significant at $p=0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	BIENNIAL	OTHER	P- value
ACRHOR	0	4	0.42
ACTLUC	12	2	0.56
ALOORE	1	14	0.05*
ALOTAI	1	7	0.19
BELAUR	6	29	0.02*
BELZOC	6	6	1.00
CASCAS	13	17	0.75
CATCLO	3	10	0.34
CATFLO	6	11	0.65
COLELE	28	2	0.11
CUPCIS	3	3	1.00
DANCHR	25	18	0.69
DINBOW	2	1	1.00
EURBRI	11	5	0.60
HARNOQ	5	0	0.59
HYAEPO	0	5	0.23
ORAARI	2	0	1.00
ORALAC	0	19	0.02*
ORASUB	2	0	1.00
PAPDEM	0	19	0.01*
PAPNIR	3	0	1.00
PARPUN	6	1	0.63
PREHIE	0	11	0.06
PREOCT	10	25	0.24
PSEVAR	1	33	0.00*
STYWIC	5	2	0.67
VANCAR	20	1	0.20

APPENDIX 2. Table 8. The butterfly species response to disturbance at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	LOW	HIGH	P- value
ACRHOR	3	0	1.00
ACTLUC	11	3	0.75
ALOORE	9	0	0.44
ALOTAI	6	0	0.60
BELAUR	21	1	0.17
BELZOC	6	7	1.00
CASCAS	16	12	0.83
CATCLO	8	2	0.62
CATFLO	3	27	0.01*
COLELE	13	14	0.97
CUPCIS	2	9	0.47
DANCHR	14	36	0.10
DINBOW	4	0	0.74
EURBRI	8	10	0.83
HARNOQ	4	0	0.76
HYAEPO	1	0	1.00
ORAARI	1	0	1.00
ORALAC	10	0	0.33
ORASUB	0	6	0.18
PAPDEM	5	2	1.00
PAPNIR	0	5	0.37
PARPUN	9	0	0.42
PREHIE	3	0	1.00
PREOCT	21	10	0.60
PSEVAR	15	2	0.38
STYWIC	1	10	0.13
VANCAR	1	29	0.03*

APPENDIX 2. Table 9. The butterfly species response to season of burn at the 10 study sites (ISA value = P-value). *= Significant at $p = 0.05$. Refer to Table 8 for butterfly species codes.

SPP. CODE	SPRING	WINTER	P- value
ACRHOR	0	5	0.27
ACTLUC	1	31	0.00*
ALOORE	5	0	0.53
ALOTAI	5	0	0.56
BELAUR	5	10	0.82
BELZOC	8	6	0.81
CASCAS	24	6	0.35
CATCLO	5	4	1.00
CATFLO	13	1	0.28
COLELE	6	35	0.01*
CUPCIS	2	6	0.52
DANCHR	18	25	0.59
DINBOW	4	0	0.79
EURBRI	2	19	0.06
HARNOQ	2	2	1.00
ORAARI	0	5	0.26
ORALAC	7	1	0.57
ORASUB	2	0	1.00
PAPDEM	5	0	0.55
PAPNIR	4	0	0.61
PARPUN	0	23	0.00*
PREHIE	2	0	1.00
PREOCT	25	6	0.24
PSEVAR	22	0	0.05*
STYWIC	5	2	0.83
VANCAR	12	8	0.81