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The effect of bushcutting on the mesic proteiod fynbos of the Riversdale coastal plain.

by Jaana-Maria Ball

THE EFFECT OF BUSHCUTTING ON

THE MESIC PROTEIOD FYNBOS

OF THE RIVERSDALE COASTAL PLAIN.

by

Jaana-Maria Ball

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ABSTRACT

Leucadendron galpinii-dominated Mesic Proteiod Fynbos on the Riversdale coastal plain recently disturbed by bushcutting was compared to undisturbed stands of similar vegetation type. In this study, it was tested whether bushcutting leads to species loss by comparing plant species numbers and cover in bushcut vegetation with non-bushcut vegetation. Traits of species in bushcut and non-bushcut vegetation were compared to determine processes most effected by bushcutting.

There was no significant differences between non-bushcut and bushcut floras in the frequency of species belonging to the different distribution, dispersal distance, dispersal mode pollination syndrome, growth form, shrub fire response and shrub leaf size categories. This was probably due to the vegetation types being accessible to similar dispersal vectors and pollinators, and certain species not being selectively bushcut. There were, however, many important differences in the frequency of individuals belonging to the different categories. These include long distance dispersal of propagules, species with unspecialized pollinators, geophyte and forb growth forms and resprouting shrubs being favoured in bushcut vegetation.

There was, however, a significant difference between the non-bushcut and bushcut floras in the frequency of species belonging to the different plant height categories. This was probably due to certain plant height categories being eliminated as a direct consequence of bushcutting. There were significant differences between the non-bushcut and bushcut floras in the frequency of *Thamnochortus insignis* individuals belonging to the different mortality and tussock diameter categories. This is probably as a result of death of individuals as a consequence of bushcutting and expansion of individuals in the uncrowded habitat. The sex of *T. insignis* individuals was not significantly different in bushcut compared to non-bushcut floras. This was probably due to male and female *Thamnochortus* individuals not being selectively bushcut.

The total number of species in the bushcut compared to the non-bushcut flora was much lower. This probably results from the creation of a more homogenous habitat in the bushcut compared to the non-bushcut vegetation, following bushcutting. The number of plots with seedling presence and the total number of seedling counts were lower in the bushcut compared to the non-bushcut floras. This is due to a decline in reproductive output in the form of seeds in bushcut vegetation or the decrease in 'propagule rain' from surrounding vegetation as a consequence of isolation.

Species diversity indexes were slightly lower in the bushcut compared to the non-bushcut flora. Species most vulnerable to extinction in bushcut vegetation were locally endemic, tall shrubs with short distance dispersal type and specialized pollinators.

The results suggest that contol of bushcutting or the provision of reseves is needed to avoid species losses.
The results of this study are likely to be locality specific. They do, however, improve our understanding of the responses of different plant types to bushcutting disturbance and recognise the need for conservation of floral diversity of the Riversdale coastal plain.

1. INTRODUCTION

The harvesting of thatch is an important veld-based industry in the agriculturally marginal areas of the southern Cape coast (Linder, 1990). The most important thatching reed species, *Thamnochortus insignis*, has a consistently high foliage projective cover (10-70%) in certain Proteiod fynbos types on the Riversdale coastal plain (Rebelo *et al.*, 1991). The demand for good quality thatch, both locally and overseas, has increased dramatically over the past few years and the cultivation of thatch species has become lucrative. This has lead to experimentation with different veld managment techniques to improve production on the Riversdale coastal plain. The practice of selectively eliminating the proteiod fynbos shrubs or bushcutting entire stands to increase thatch yields has become increasingly popular. As a result the vegetation types of the area with a high cover of thatch (*T. insignis* and *T. erectus*) are becoming increasingly disturbed and fragmented.

The coastal forelands of the fynbos biome are characterized by high floristic and vegetational complexity (Boucher, 1987). Lowland Fynbos shrublands of the south-western Cape include many hundreds of highly localized endemics (Cowling, 1990; Cowling and Holmes, in press; Cowling *et al.*, in press). The flora of the Riversdale coastal plain has been poorly-studied compared to other regions in the fynbos biome. The only major ecological work undertaken in the area is that of Muir (1929) who described the Riversdale coastal vegetation. Recently the vegetation has been structurally classified and described by Rebelo *et al.* (1991). Using limited floristic data (dominant species) they recognized twenty major vegetation categories in the area. A field guide for the area has also recently been published (Bohnen, 1986). Species richness of the Riversdale coastal plain (*ca.* 1 580 spp. in 2 800 km^2) is lower than that of the Agulhas plain (*ca.* 1 800 spp. in 1 500 km^2) (Cowling *et al.* 1988). Fifty-four per cent (1 100 km^2 of the 2 030 km^2) of the Limestone Fynbos and 88% (420 km^2 of the 480 km^2) of the Dune Fynbos occurs in and immediately adjacent to the Riversdale coastal plain (Rebelo *et al.*, 1991). The Riversdale coastal plain contains the largest development of Tertiary limestone in the Fynbos Biome and the largest Enon deposits on the south coast. Conservation planning in the region is particularly lacking (Jarman, 1986), and levels of cultivation, overgrazing, bushcutting and alien infestation are rapidly increasing and they are associated with the deterioration of the natural vegetation. As a result, many of the highly localized endemics (Hall and Veldenhuis, 1985) have become confined to small habitat fragments as a result of extensive land transformation (Hilton-Taylor and Le Roux, 1989). Numerous authors have expressed recognised the urgent need for greater protection of lowland Fynbos (Rebelo and Siegfried, 1990; Taylor, 1978; Boucher and Moll, 1980).

Leucadendron galpinii-dominated Proteoid Fynbos is endemic to the Riversdale coastal plain and has the highest concentration of *Thamnochortus insignis* (Rebelo *et al.*, 1991), the most desirable thatch. Thus, disturbance (bushcutting) in this vegetation type could result in the extinction of certain species, including local endemics.

This aim of this study was to access the effects of bushcutting on the proteiod fynbos vegetation, and to determine which processes and species groups are most sensitive to bushcutting. Of particular interest was how habitat destruction by bushcutting affects extinction of rare local endemics.

As pointed out by Cowling and Bond (1991) the problem with a comparitive approach, such as the one used in this study, is that it places heavy emphasis on the total species richness and none on the identity of the species that survive bushcutting.

A comparison of the bushcut and non-bushcut floras was undertaken to answer the following questions: Does bushcutting result in plant species loss? and Which species disappear as a result of bushcutting?

2. STUDY AREA

The study area is on the Riversdale Plain (2 800 km^2) (**Figure 1**), which forms part of the south-western Cape coastal forelands. The plain is defined as the coastal plain south of the Langeberg Mountains between the Duiwenhoks River in the west and the Gouritz River in the east. The Mio-Pliocene limestones and associated colluvial deposits of the Bredasdorp formation form distinctive relief features in the coastal zone. The study site is the farm Wolwekop, 20 km south of Riversdale ($34^{o}15'25"$S, $21^{o}13'10''$) and is situated on a level plain (flat to 10^{o}) at 120 m above sea level between two limestone dune crests. Colluvial soils which overlay the limestone to a depth greater than 1 m have a 0.1-0.4 m Orthic A horizon over deep structureless brown/grey sand (Fernwood langebaan) and are of moderate alkalinity (pH 7.4) (Rebelo *et al.*, 1991). The habitat does not appear to be seasonally waterlogged.

The climate of the area is relatively uniform (Fuggle, 1981). Mean annual rainfall is 426 mm at Riversdale (20 km south of the study site) and 430 mm at Stilbaai (30 km south-east of the study area). Rainfall seasonality is typical of a mediterranean-type climate with between 26 and 29% of the annual precipitation falling in the winter months (May to October), 15-19% in summer (November-March), with a peak rainfall period in the autumn months (March-May) (28-31%). Temperatures range from a mean daily summer maximum of 22.0^{o}C, averaging betweeen 17.1 and 17.4^{o}C for the year. Frost is extremely rare and snow falls have not been recorded (Weather Bureau, 1986).

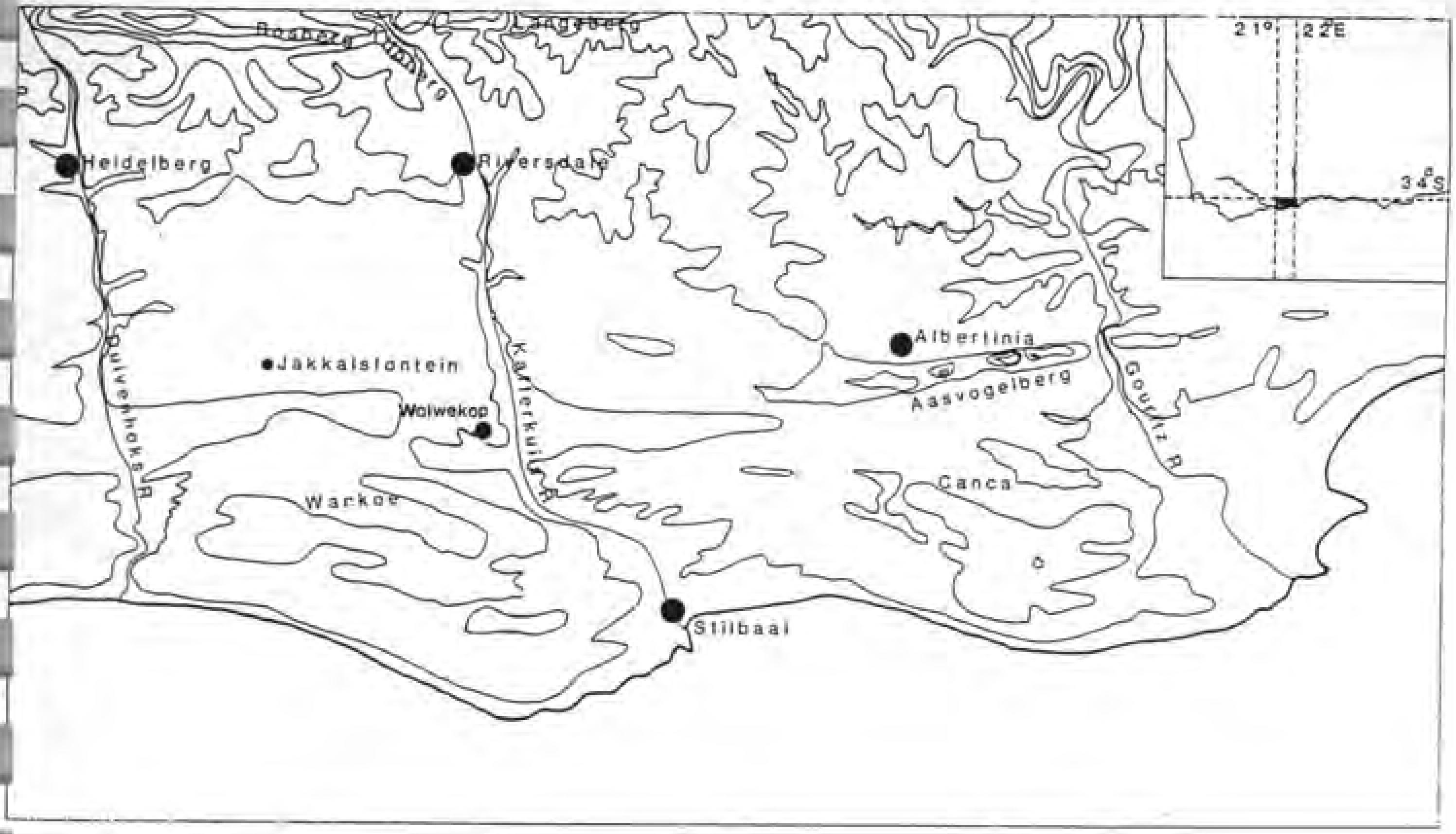


Fig. 1. Map of the Riversdale Plain showing the location of the study site, Wolwekop (Adapted fom Rebelo *et al.* 1991)

The Riversdale Plain falls within the Cape Floristic Region (Bond and Goldblatt, 1984) and is dominated by fynbos, an evergreen sclerophyllous, heath-like shrubland associated with nutrient poor soils. The shrublands are fire-prone and usually burn at six to 40 year intervals (Cowling *et al.* 1987; Bond, 1984). The predominant vegetation in the study area (**Figure 2**) is Mesic Proteiod Fynbos [>10 % cover of >1 m tall non-sprouting Proteaceae with a high cover of Restiod (10-70%, with >5% cover in the 1-2 m stratum) and low cover of Ericaceous Fynbos understorey] (Campbell, 1985; Rebelo *et al.*, 1991). *Thamnochortus insignis* is the dominant restiod interspersed in the fynbos. The vegetation subseries is further classified to the *Leucadendron galpinii* type, based on the dominant proteoid species. The vegetation type is confined to the Riversdale coastal plain (Williams, 1972). The non-bushcut vegetation was approximately 15 years of age.

Fig. 2. Non-bushcut (above) and bushcut (below) vegetation at the study site, Wolwekop on the Riversdale Plain (photo: R.M. Cowling)

3. METHODS

3.1. Data collection

The research site was chosen because it fulfilled the following conditions:

(i) A fence, approximately 1.5 km long, separates the same fynbos communities under different bushcutting regimes (bushcut and non-bushcut).

(ii) The area is topographically and edaphically fairly uniform; in so far as the fence cuts across habitat boundaries and does not run parallel to them.

(iii) The fence is known to have separated areas with consistently different bushcutting treatments for several years before the beginning of the study.

Both sites had been lightly grazed and the *T. insignis* harvested in March of the study year. The bushcut site had been bushcut (after harvest and removal of the thatch) for the first time five years previous to the study and again a year previous to the study (personal communication with Mr P. de Kok). Bushcutting was performed machanically by a tractor, set at a cutting height of approximately 30 cm.

In this study we use a similar approach as Bond *et al.* (1988) and Cowling and Bond (in press) for determining the existence of, and the processes responsible for, fragmentation effects in Fynbos shrublands in the southern and south-western Cape lowlands, respectively.

The vegetation was recorded in twelve pairs of adjacent, systematically located plots of 25 m^2 (5 m x 5 m) in the non-bushcut and bushcut vegetation, respectively. Plots (on the same side of the fence) were placed at a distance from each other to ensure that the samples were representative of the entire community. The pairs of samples were taken in such a way that complication of bushcutting treatment effects by habitat heterogeneity was minimized. The paired plots on either side of the fence were similar in all respects except for the effects of bushcutting. Spatial effects at the fence itself may affect results (cattle tracks, truck tracks, massive seed dispersal across the fence). Plots were removed 10 m from the fence to reduce these. Places where there was an obvious differences in habitat (rockiness, slope) across the fence were avoided.

Data were collected over 15 days in the months of February, April, May and June 1991. A plant specimen collection trip in September (spring) of the same year ensured that most of the ephemeral species (deciduous geophytes and annuals) were collected. Species lists were compiled by noting all vascular plants encountered in the plot and the relative contribution of each species to foliage projective cover (percentage) was subjectively estimated. Percentage cover of less than 1% was noted as <1% and given the value of 0.5% in further calculations. The height of each plant species encountered in the quadrat was subjectively estimated. The two plots belonging to a pair were recorded on the same day by the same observer. Species nomenclature follows Bond and Goldblatt (1984).

3.2. Data analysis

Each species encountered in the survey was allocated to categories of distribution range, dispersal distance, dispersal mode, pollination syndrome, growth form and plant height (potential plant height). Distribution range categories included local endemics (confined to the Riversdale Plain), regional endemics (confined to the Cape Floristic Region) and non-endemics (widely distributed outside these centres). Shrubs were further categorized into seeder or sprouter fire response categories and leaf size categories. The other categories are self explanatory in **Table 3**. Data were obtained from Bond and Goldblatt (1984), Bond and Slingsby (1983), Rebelo (1987), Raunkiaer (1934), taxonomic monographs and my own personal observations, as well as those of R.M. Cowling, W.J. Bond and E.G.H. Oliver. Individuals of *T. insignis* were allocated to categories of tussock diameter (measured as the apparent mean diameter of an individual at ground level) and plant sex. It was also noted whether individuals were dead or living. Differences in the frequencies of these categories in the non-bushcut and bushcut floras were tested by drawing up contingency tables. Chi-squared analysis was used to test the null hypothesis that the number of species in each category for the bushcut flora would be distributed in the same proportion as the not bushcut flora. To obtain sufficient numbers in each cell of the contingency table, >1 m and >2 m plant species height, passive and ballistic dispersal type, succulent and shrub growth form and forb and climber growth form categories were grouped. For species exclusive to the non-bushcut and bushcut floras, frequencies were below the minimum group size of five. Chi-squared analyses were performed with a minimum group size of <5 and as a result significance levels could not be calculated.

The floristic relationships between the non-bushcut and bushcut vegetation data were analysed using detrended correspondence analysis (Hill, 1979). Plots three (non-bushcut and bushcut) were omitted from the analysis because they did not share many species with other plots. Eigenvalues, indicating the importance of the ordination axes, were 0.31357 and 0.24354 for axis 1 and axis 2, respectively (Ter Braak, 1987). Indirect gradient analysis using detrended correspondence analysis is useful because it avoids the problem of the "arch" or "horseshoe" effect due to the quadratic dependency of the second axis on the first axis, and the compression of the axis ends (Hill, 1979; Magurran, 1988). Detrending, however, also attempts to impose such a homogeneous distribution of scores on the data where none exist (Ter Braak, 1986).

Diversity indexes were calculated using percentage cover data of the species in each of the 12 plots of the two vegetation types. The Simpson's and Shannon-Weiner indexes were calculated to determine whether there were any differences in the species diversity between the two vegetation types. The equation used to calculate Simpson's index is

$$D = \frac{(n_i\ (n_i - 1)}{(N\ (N - 1)}$$

where n = the percentage cover of individuals in the ith species, and N = the total percentage cover of individuals.

The formula used for calculating the Shannon index is

$$H' = - \ p_i \ln p_i$$

where p_i, the proportional percentage cover of the ith species = (n_i/N).

The eveness of the two vegetation types were calculated using the formula

$$E = H'/\ln S$$

The Berger-Parker index was calculated to determine whether there is any difference in the dominance of plant species between the two vegetation types. The Berger-Parker index is calculated from the following equation

$$d = N_{max}/N$$

where N = total percentage cover of individuals and N_{max} = percentage cover of the species with the highest percentage cover value.

Reciprocal forms of the Simpson's and Berger-Parker indexes were calculated for convenience. The mean and standard deviation were calculated for each index.

As the paired plots were similar in all respects (i.e. topographically and edaphically), except for the bushcutting treatment, the plots were treated independently and inferential statistics were used. The paired *t*-test was used to determine whether, in paired plots, the number of times a value was higher on the one side of the fence was significantly greater than the number of times it was higher on the other side. A Hewlett-Packard scientific calculator (HP-41CX) was used to calculate the *t*-statistic and the probability value (*p*) was obtained from a probability table (Zar, 1984).

4. RESULTS

4.1. Patterns of species richness in relation to bushcutting

4.1.1. Changes in species richness

Frequency values

Table 1. presents the data on species diversity in the bushcut and non-bushcut sites. For comparable sample areas, bushcut vegetation had fewer species than the non-bushcut vegetation. Nine percent of the non-bushcut vegetation consisted of rare local endemics compared to seven percent of the bushcut vegetation. The bushcut vegetation also had fewer species exclusive to the treatment. There appears to be little recruitment through seedling establishment in the bushcut site, compared to the non-bushcut site (**Table 2**). All diversity index values were lower for the bushcut compared to the non-bushcut floras, although the difference was not significant for the number of species/25 m^2 and the Equitability index. Simpson's and Shannon-Weiner diversity indexes indicated a significant decrease in species diversity following bushcutting. Species were found to be more equally abundant in the non-bushcut vegetation compared to the bushcut vegetation. The bushcut vegetation was found to have a greater degree of species dominance compared to the non-bushcut vegetation, as indicated by the Berger-Parker and Simpson's indexs.

TABLE 1

Species diversity indexes for non-bushcut and bushcut plots (25 m^2) in Proteiod Fynbos on the Riversdale Plain (n = 12 for each treatment). Non-count data are mean ± standard deviation

	Non-bushcut (Control)	Bushcut	Paired t-statistic
Total number of species	111	88	
Total no. of species exclusive to the treatment	36	13	
No species/25 m^2	30.08 ± 3.53	26.92 ± 5.98	1.84NS
Simpson's index[1]	8.81 ± 8.71	5.84 ± 8.58	2.96**
Equitability E (D/S)	0.29 ± 0.26	0.20 ± 0.23	2.21*
Shannon-Weiner index	2.36 ± 0.39	1.83 ± 0.5	4.06***
Equitability E	0.59 ± 0.21	0.53 ± 0.15	0.94NS
Berger-Parker index[1]	3.47 ± 1.9	2.24 ± 1.48	2.77*

[1]Reciprocal forms of the Simpson'and Berger-Parker indexes are displayed.
* p = <0.05. ** p = <0.01. *** p = < 0.001. NS not significant

TABLE 2

Seedling presence, total percentage cover and percentage cover of local endemics for non-bushcut and bushcut plots (25 m^2) in Proteoid Fynbos on the Riversdale Plain (n = 12). Non-count data are mean ± standard deviation

	Non-bushcut (Control)	Bushcut	Paired *t*-statistic
Seedling presence (number of plots)	11	8	
Seedling presence (no. of counts)	25	16	
Total percentage cover	81.83 ± 21.01	66.45 ± 23.36	2.36*
Percentage cover of local endemics	41.58 ± 24.27	16.38 ± 10.43	3.14**

* p = <0.05. ** p = <0.01. *** p = < 0.001. NS not significant

The bushcut vegetation showed qualitative changes from a floristic point of view. The main dominant species in the non-bushcut site was *Leucadendron galpinii*. Many shrub species such as *Metalasia muricata*, *Aspalathus* sp., *Nylantia spinosa*, *Chrysocoma ciliata*, *Protea repens*, *Phylica* sp., *Agathosma riversdalensis* and *Passerina galpinii* are common in the non-bushcut site.

The main dominant species in the bushcut site is the restio *Thamnochortus insignis* and perennial grass *Cynodon dactylon*.

The low reprouting shrub *Ifloga repens* was also a dominant in one bushcut quadrat.

Ordination data

Detrended correspondence analysis (Hill, 1979) on all the plots separates all the bushcut from the non-bushcut plots indicating distinct compositional differences between the two vegetations (**Figure 3a**). The ordination shows how the plots relate to the steep disturbance gradient across the fenceline. In relation to the control sites (non-bushcut sites), bushcutting limited the variation in floristic composition to a relatively smaller domain in multivariate space characterised by high amounts of *Thamnochortus insignis*, *Cynodon dactylon*, *Oxalis luteola*, *Oxalis ciliaris*, *Tephrosia capensis*, *Cyanchum obtusifolium*, *Acacia saligna*, *Chironia baccifera*, *Lachenalia rubida*, *Solanum quadrangulare*, and 'forb hairy-vein' (**Figure 3b**). From the ordination it is evident that bushcutting disturbance is sufficient to predict the main part of the variation in the species composition (Ter Braak, 1986).

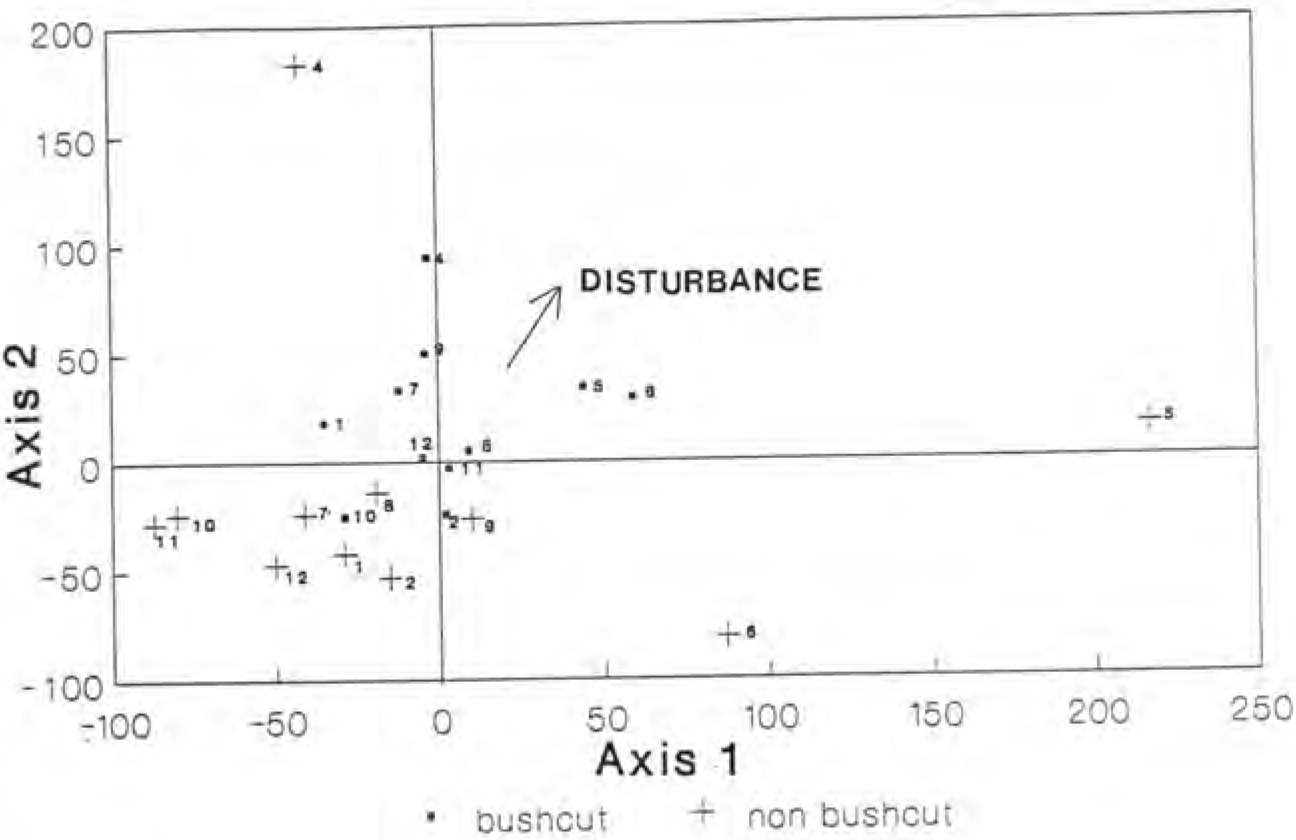
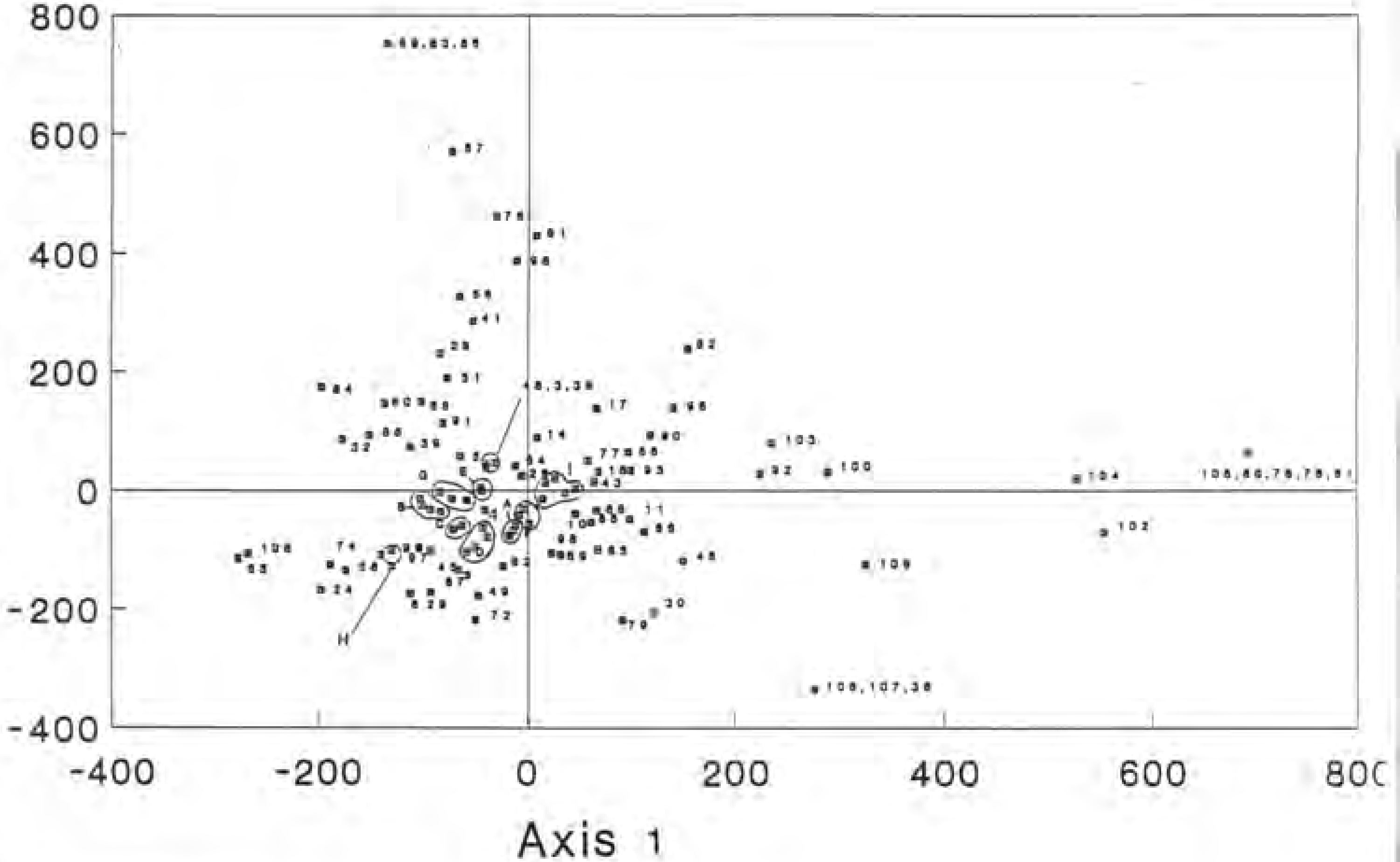


Fig. 3a. Detrended Correspondence Analysis ordination of the 12 non-bushcut and 12 bushcut plots utilizing vascular plant species cover abundance. Species were weighted by cover into five classes: <1%, 1-5%, 6-25%, 26-50%, 51-75% and >75%.



KEY FOR ORDINATION

A = Species 21, 42 and 94
B = Species 6, 14, 37 and 71
C = Species 12 and 96
D = Species 23, 27, 49 and 101
E = Species 7 and 19
F = Species 13, 25 and 94
G = Species 20, 53 and 44
H = Species 9, 47 and 70
I = Species 2, 13, 15, 33 and 62

Fig. 3b. Detrended Correspondence Analysis ordination of the 123 plant species in non-bushcut and bushcut plots utilizing vascular plant species cover abundance. Species were weighted by cover into five classes: <1%, 1-5%, 6-25%, 26-50%, 51-75% and >75

KEY TO PLANT SPECIES

1 Leucadendron galpinii E. Phillips Hutch.
2 Thamnochortus insignis Masters
3 Rhus glauca (Thunb.)
4 Chrysocoma ciliata Berg.
5 Tephrosia capensis (Jacq.)
6 Anthospermum aethiopicum (L.)
7 Pentaschistis thunbergii (Kunth) Stapf
8 Metalasia muricata (L.)
9 Phylica humilis Sonder
10 Protasparagus capensis (L.)
11 Pelargonium triste (L.)
12 Karroochloa curva (Nees)
13 Passerina vulgaris Thoday
14 Cynadon dactylon (L.)
15 Microloma sagittum (L.)
16 Apiaceae
17 Aspalathus submissa R. Dahlgren
18 Selago sp.
19 Massonia pustulata Jacq.
20 Eragrostis curvula (Shrader) Nees
21 Ehrharta villosa Schultes f.
22 Platycalyx pumila N.E.Br.
23 Crassula cymosa Bergius
24 Podalyria myrtillifolia Willd.
25 Galenia herniariaefolia (Presl) Fenzl
26 Lachenalia rubida Jacq.
27 Senecio
28 Helichrysum crispum (L.)
29 Haemanthus coccineus L.
30 Stipagrostis zeyheri subspp. macropus (Nees)
31 Hermannia decumbens Willd. ex Spengel.
32 Agathosma riversdalensis Dummer
33 Indigofera heterophylla Thunb.
34 Oxalis ciliaris Jacq.
35 Cyanella hyacinthoides L.
36 Gnidia sp.
37 Helichrysum asperum (Thunb.)
38 Osteospermum sp.
39 Heliophila subulata Burchell ex DC.
40 Lampranthus glaucus (L.)
41 Tetragonia decumbens Miller
42 Carpobrotus muirii (L. Bolus)
43 Nylantia spinosa (L.)
44 Oxalis luteola Jacq.
45 Helichrysum cochleariforme DC.
46 Ischyrolepis sieberi (Kunth)
47 Tetraria cuspidata (Rottb)
48 Clutia ericoides Thunb.
49 Solanum hermannii Dunal
50 Geophyte onion-like
51 Ornithogalum juncifolium Jacq.
52 Ehrharta calycina Smith
53 Forb hairy vein
54 Anomalanthus scoparius Klotzsch.

55 Ifloga repens (L.)
56 Pelargonium longicaule spp. convolvulifolia (Jacq.)
57 Phylica stipularis L.
58 Dorotheanthus gramineus (Haw.) Schwantes
59 Cliffortia tuberculata (Harvey) Werm.
60 Chironia baccifera (L.)
61 Nemesia versicolor E. Meyer ex Benth.
62 Protasparagus retrofractus (L.)
63 Lobelia capillifolia (Presl) A. DC.
64 Acacia saligna (Labill.)
65 Arctotis candida Thunb.
66 Ficinia dunensis Levyns
67 Othonna filicaulis Jacq.
68 geophyte 2
69 Ficinia tristachya (Rottb.) Nees
70 Phylica ericoides (L.)
71 Drosanthemum floribundum (Haw.) Schwantes
72 Atriplex sp.
73 Ficinia lateralis (Vahl) Kunth
74 Aspalathus simii H. Bol. ssp. katbergensis Dahlg.
75 Disparago kraussii Sch. Bip.
76 Geophyte paired leaf
77 Bulbine filifolia Baker
78 Relhania squarrosa (L.) L'Herit.
79 Passerina galpinii C.H. Wright
80 Struthiola sp.
81 Koeleria capensis (Thunb.) Nees
82 Aspalathus nigra (L.)
83 Leucadendron salignum Bergius
84 Ficinia truncata (Thunb.) Schrader
85 Thamnochortus arenarius Esterh. ined.
86 Pharnaceum dicotomum L.f.
87 Babiana tubulosa (Burm.f.) Ker-Gawl. var. tubiliflora (L.f.) G.J. Lewis
88 Protasparagus africanus (Lam.) Oberm.
89 Aizoon sp.
90 Babiana nana (Andr.) Spreng. var. angustifolia (Eckl.) G.J. Lewis
91 Hypodiscus procurrens Esterh. ined.
92 Themeda triandra Forsskal
93 Phylica parviflora Bergius
94 Helicrysum dasyanthemum (Willd.) Sweet
95 Gazania sp.
96 Felicia minima (Hutch.)
97 Helichrysum teretifolium (L.) D. Don.
98 Phylica rubra Willd. ex Roerner and Schultes
99 Lightfootia nodosa Buek
100 Grass-unidentified
101 Oxalis obtusa Jacq.
102 Cyanchum obtusifolium L.f.
103 Oxalis polyphylla Jacq.
104 Massonia echinata L.f.
105 Hesperantha falcata (L.f.) Ker Gawler
106 Rhus laevigata L.Saldanha
107 Pelargonium spp.
108 Hellmunthia membranacea (Thunb.) R. Hainesand K. Lye
109 Struthiola sp.

110 Microloma tenuifolium (L.) K. Schum.
111 Metalasia gnaphaloides (Thunb.) Druce
112 Oxalis smithiana Ecklon and Zeyher
113 Solanum quadrangulare Thunb. ex. L.f.
114 Cissampelos capensis L.f.
115 Rumex sp.
116 Protea repens (L.)
117 Acacia cyclops A. Cunn. ex G. Don
118 Hermannia diversistipula Presl. ex. Harvey
119 Erica discolor Andr.
120 Simocheilus dispar N.E.Br.
121 Cliffortia stricta Werm.
122 Cliffortia falcata L.f.
123 Ehrharta ramosa (Thunb.) subsp. aphylla

Species nomenclature follows Bond and Goldblatt (1984), Bohnen (1986) and the Bolus Herbarium.

Table 3. indicates the distribution of species traits. There was no significant differences between non-bushcut and bushcut floras in the frequency of species belonging to the different distribution, dispersal distance, dispersal mode, pollination syndromes, growth forms, shrub fire response and shrub leaf size categories. Although not significant, there were noticable changes in the frequency of individuals belonging to the categories described above between the non-bushcut and bushcut vegetation. Non-bushcut vegetation had a higher frequency of regional endemics, ant dispersed species, passive dispersed species and ballistic dispersed species, although these were not statistically significant. Bushcutting resulted in a significant decrease in the frequency of species with short and medium distance dispersal and a slight increase in the frequency of species with long distance dispersal. Bushcut plots showed a marked decrease in the frequency of shrubs. A higher percentage of the shrubs were sprouters in the bushcut compared to non-bushcut vegetation. Bushcut vegetation showed a decrease in the frequency of insect, wind and bird pollinated species, although the largest decrease was for insect pollinated species.

There was however a significant difference between the non-bushcut and bushcut floras in the frequency of species belonging to the different plant height categories. The frequency of individuals in the lowest plant height category increased in the bushcut samples and less than 5% of all the individuals were found in height categories greater than 50 cm.

TABLE 3

The distribution of species traits of non-bushcut and bushcut plots (25 m^2) in Proteoid Fynbos on the Riversdale Plain. (n = 12 for each treatment). Non-count data are mean ± standard deviation

	Non-bushcut (Control)	Chi-squared	Bushcut	Species exclusive to non-bushcut (Control)	Chi-squared	Species exclusive to bushcut
Distribution range						
Local endemic	10 (9)		7 (8)	4 (11)		1 (8)
Regional endemic	63 (57)		51 (58)	19 (53)		7 (54)
Widespred	25 (23)		19 (22)	8 (22)		2 (15)
Unknown	13 (12)		11 (13)	5 (14)		3 (23)
		0.16NS			0.32[1]	
Dispersal distance						
Short (0-10 cm)	27 (24)		17 (19)	12 (33)		2 (15)
Medium (11-50 cm)	73 (66)		59 (67)	1 (3)		10 (77)
Long (>50 cm)	11 (10)		12 (14)	23 (64)		1 (8)
		2.16NS			265.82[1]	
Dispersal mode						
Wind	73 (65)		59 (67)	23 (64)		10 (76)
Ant	12 (11)		7 (8)	6 (17)		1 (8)
Ballistic	12 (11)		9 (10)	4 (11)		1 (8)
Vertebrate	10 (9)		11 (13)	1 (3)		1 (8)
Passive	4 (4)		2 (2)	2 (5)		0 (0)
		2.10NS			1.09[1]	
Pollination syndrome						
Wind	32 (29)		22 (25)	9 (25)		1 (8)
Insect	76 (68)		65 (74)	25 (69)		12 (92)
Bird	3 (3)		1 (1)	2 (6)		0 (0)
		1.19NS			2.3[1]	
Growth form						
Shrub	48 (43)		35 (40)	18 (50)		4 (31)
Graminoid	21 (19)		14 (16)	7 (19)		1 (8)
Forb	9 (8)		10 (11)	2 (6)		3 (22)
Geophyte	16 (14)		15 (17)	5 (14)		4 (31)
Creeper	4 (4)		5 (6)	0 (0)		1 (8)
Succulent	4 (4)		3 (3)	1 (3)		0 (0)
Annual	9 (8)		6 (7)	3 (8)		0 (0)
		3.70NS			3.70[1]	
Fire response (shrubs only)						
Sprouter	17 (35)		15 (42)	3 (17)		1 (25)
Seeder	31 (65)		21 (58)	15 (83)		3 (75)
		0.61NS			0.2[1]	
Shrub leaf size						
Leptophyll	36 (73)		25 (71)	15 (83)		3 (100)
Nanophyll	13 (27)		10 (29)	3 (17)		0 (0)
		0.07NS			—[2]	
Plant height (cm)						
0-25	62 (56)		46 (52)	20 (55)		11 (84)
26-50	28 (25)		22 (25)	11 (31)		1 (8)
51-100	11 (10)		12 (14)	3 (8)		1 (8)
101-200	8 (7)		6 (7)	2 (6)		0 (0)
>200	2 (2)		2 (2)	0 (0)		0 (0)
		65.06***			3.9[1]	

Values in brackets refer to percentages of species in categories. * p = <0.05. ** p = <0.01. *** p = < 0.001. NS not significant. [1] significance not tested. [2] too few variables for chi-squared to be tested

An method of testing for vulnerability to extinction is to compare traits of species found only in the non-bushcut samples with species found only in the bushcut samples. As significance levels could not be calculated, qualitative changes in the frequency of species belonging to the different categories found exclusively in non-bushcut and bushcut vegetation samples will be discussed.

Table 4. shows a list of 36 species and their traits that are exclusive to the non-bushcut site. **Table 5.** shows a list of 13 species and their traits that are exclusive to the bushcut site. **Table 3** indicates the distribution of species traits. Bushcut vegetation samples contained fewer local endemic species exclusive to the treatment. Two of the endemics found in the non-bushcut vegetation samples were non-sproutng Ericaceae and the other two graminoides. The majority of species exclusive to bushcut plots had medium distance dispersal, compared to the non-bushcut plots where the majority had long and short distance dispersal. The majority of species exclusive to bushcut plots were wind dispersed and the frequency of ant, ballistic, vertebrate and passive dispersed species exclusive to the bushcut site was less than that for the non-bushcut site, although the percentage of vertebrate dispersed species increased following bushcutting. The majority of species exclusive to bushcut plots were insect pollinated, with very few species being bird or wind pollinated compared to the non-bushcut vegetation. The majority of species exclusive to bushcut vegetation samples belonged to forb and graminoid growth form categories, whereas the in the

TABLE 4

List of species traits of those species confined to the non-bushcut Proteoid Fynbos samples (n=12) at Wolwekop on the Riversdale Plain. KEY: Dist=distribution, GF=growth form, FR=fire response, Hgt=plant height, DM=dispersal mode, DD=dispersal distance and P=pollination syndrome.

	Dist	GF	FR	Hgt	DM	DD	P
Acacia saligna	W	S	SR	1	V	L	I
Aizoon spp.	?	F		1	P	S	I
Aspalathus simii	C	S	SS	1	B	S	I
Atriplex sp.	?	F		1	W	M	I
Babiana tubulosa	C	GE		1	W	M	I
Bulbine filifolia	C	GE		2	B	S	I
Cliffortia stricta	C	S	SS	3	A	S	W
Disparago kraussii	C	S	SS	2	W	M	W
Dorotheanthus gramineus	C	A		1	W	M	I
Erica discolor	C	S	SR	3	W	M	B
Ficinia tristachya	C	GR		2	W	M	W
Ficinia truncata	E	GR		1	A	S	W
Filicia minima	C	A		1	W	M	I
Geophyte onion-like	?	GE		1	W	M	I
Gnidia sp.	?	S	SS	3	A	S	I
Haemanthus coccineus	C	GE		1	W	M	I
Hermannia diverstipula	C	S	SS	2	W	M	I
Hypodiscus procurrens	E	GR		1	A	S	W
Karoochloa curva	W	GR		2	W	M	I
Koeleria capensis	W	GR		1	W	M	W
Leucadendron salignum	C	S	SR	4	W	M	I
Lightfootia nodosa	W	S	SS	2	B	S	I
Nemesia versicolor	C	A		1	W	M	I
Oxalis smithiana	W	GE		1	W	M	I
Passerina galpinii	C	S	SS	2	P	S	W
Pharnaceum dicotonum	W	SU		1	W	M	I
Phylica ericoides	W	S	SS	1	A	S	I
Phylica parviflora	C	S	SS	2	A	S	I
Platycalyx pumila	E	S	SS	2	W	M	I
Podalyria myrtillifolia	C	S	SS	1	B	S	I
Protea repens	C	S	SS	4	W	M	B
Relhania squarrosa	C	S	SS	1	W	M	I
Simocheilis dispar	E	S	SS	2	W	M	I
Struthiola sp.	?	S	SS	1	W	M	I
Thamnochortus arenarius	C	GR		2	W	M	W
Themeda triandra	W	GR		1	W	M	W

TABLE 5

List of species traits of those species confined to the bushcut Proteoid Fynbos samples (n=12) at Wolwekop on the Riversdale Plain. KEY: Dist=distribution, GF=growth form, FR=fire response, Hgt=plant height, DM=dispersal mode, DD=dispersal distance and P=pollination syndrome.

	Dist	GF	FR	Hgt	DM	DD	P
Anomalanthus scoparius	C	S	SS	1	W	M	I
Cynanchum obtusifolium	W	C		1	W	M	I
Geophyte 2	?	GE		1	W	M	I
Hellmunthia membranaceae	C	GR		3	W	M	W
Lobelia capillifolia	C	F		1	B	S	I
Massonia echinata	C	GE		1	W	M	I
Metalasia gnaphaloides	C	S	SS	2	W	M	I
Osteospermum sp.	?	F		1	W	M	I
Oxalis luteola	C	GE		1	W	M	I
Oxalis polyphylla	C	GE		1	W	M	I
Phylica humilis	E	S	SS	1	A	S	I
Protasparagus retrofractus	W	S	SR	1	V	L	I
Selago sp.	?	F		1	W	M	I

KEY FOR SPECIES TRAITS

<u>Growth Form</u>	
Annual	A
Geophyte	GE
Graminoides	GR
Shrubs	S
Creeper	C
Forb	F
Succulent	SU
<u>Fire response</u> (shrubs only)	
Seeder	SS
Sprouter	SR
<u>Distribution</u>	
Widely distributed	W
Cape foristic region	C
Regional or local endemic	E
<u>Dispersal Mode</u>	
Ant dispersed	A
Wind dispersed	W
Ballistic	B
Vertebrate	V
Passive	P
<u>Dispersal Distance</u>	
Short (0-10m)	S
Medium (10-50m)	M
Long (>50m)	L
<u>Pollination Type</u>	
Insect	I
Bird	B
Wind	W
<u>Plant height</u> (cm)	
0-25	1
25-50	2
50-100	3
>100	4
>200	5

non-bushcut vegetation samples the majority of species were shrubs. The frequency of reseeding shrubs in vegetation exclusive to the two treatments was higher in non-bushcut. All the shrub species exclusive to the bushcut site had leptophyll size leaves. The majority of plant species exclusive to bushcut vegetation samples were <25 cm tall, wheras there was a fairly even spred between the plant height classes <200 cm in the non-bushcut samples.

4.1.2. Changes in species abundance

Percentage cover

Overall species composition for the bushcut and non-bushcut communities is fairly similar, but relative abundances of most species changes dramatically. Total percentage cover decreased significantly as a result of bushcutting (**Table 2.**). Mean percentage cover of local endemics in the non-bushcut plots was 2.5 times greater than that of the bushcut plots (**Table 2.**). **Figure 4.** shows percentage cover values for the different plant growth forms of the non-bushcut and bushcut sites. Percentage cover values for reseeding shrubs, climbers and succulent growth form groups is largely reduced when the community is bushcut. Percentage cover values for sprouting shrubs, graminoids, forbs, geophytes and annuals increased when the community was bushcut.

Figure 5. shows percentage cover values for the different plant height classes. It is evident that bushcutting results in

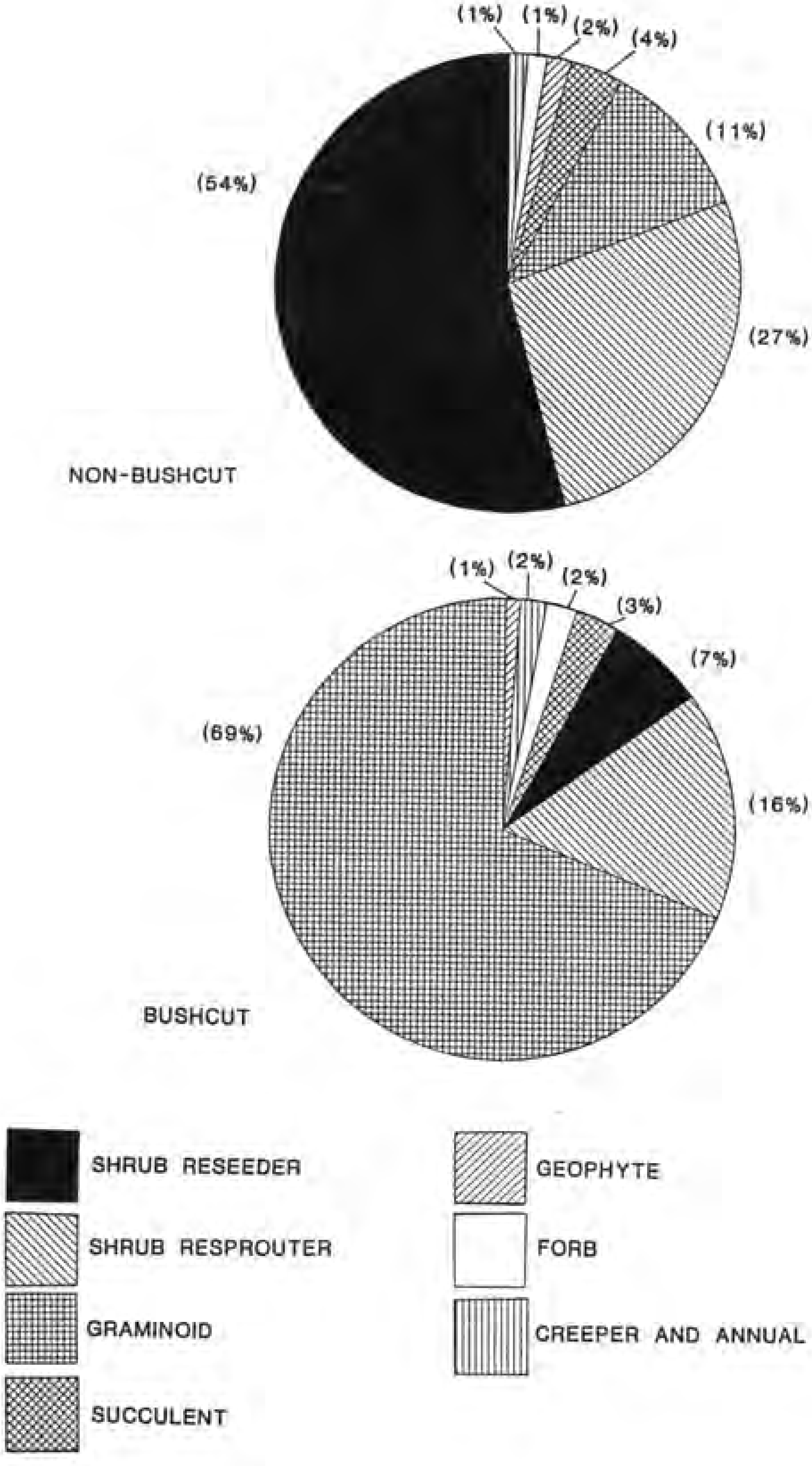


Fig. 4. Pie charts comparing percentage cover of the different growth forms of non-bushcut (n=12) and bushcut vegetation (n=12) of the Riversdale Plain

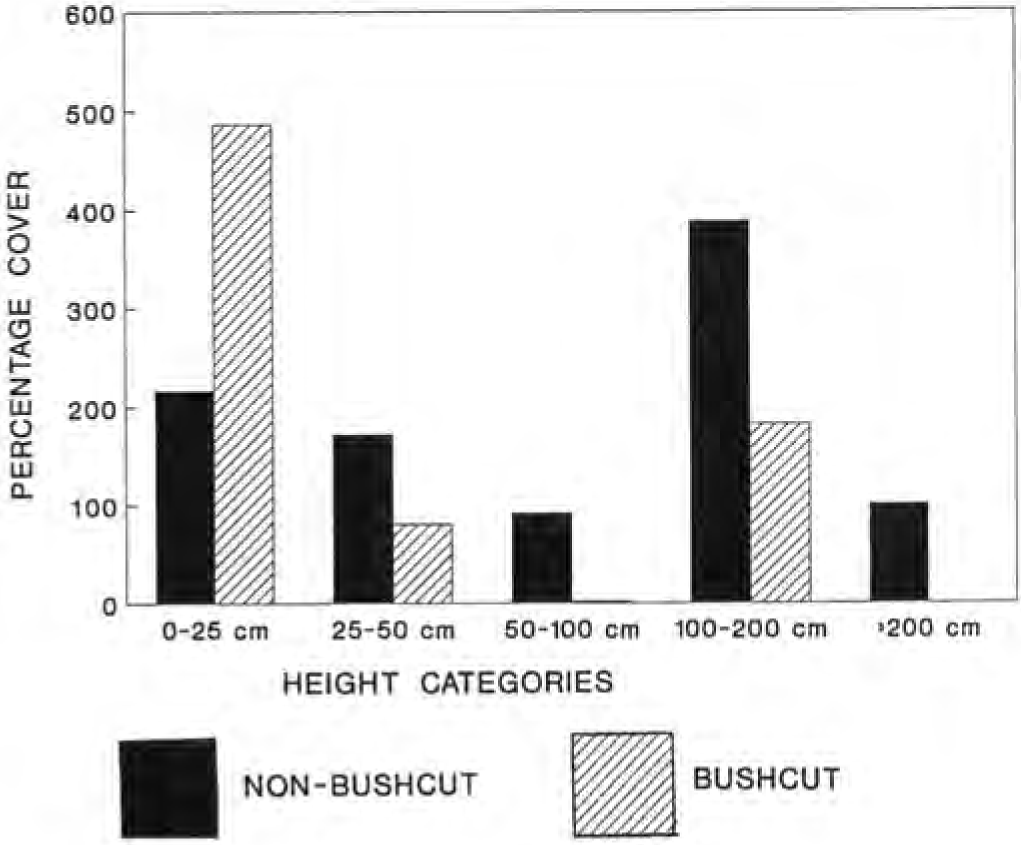


Fig. 5. Percentage cover of the non-bushcut (n=12) and bushcut (n=12) vegetation belonging to the various height class categories.

increased cover in the lowest plant height class and a reduction in cover in all higher classes.

4.2. *Thamnochortus insignis* population

Bushcutting results in an increase in the total number of *T. insignis* individuals per unit area. There was no significant difference between the non-bushcut and bushcut floras in the frequency of individuals belonging to the different sex categories. The number of reproductive individuals decreased slightly in bushcut compared to the non-bushcut sites (**Table 6.**). The number of dead individuals was shown to increase with bushcutting (**Table 6.**). The frequency of individuals in all tussock diameter categories was significantly greater in the bushcut compared to the non-bushcut site (**Table 6.**).

TABLE 6

The Distribution of *Thamnochortus insignis* traits in non-bushcut and bushcut plots (25 m^2) in Proteoid Fynbos on the Riversdale Plain. (n = 12 for each treatment). Non-count data are mean ± standard deviation

	Non-bushcut (Control)	Chi-squared	Bushcut
Number of individuals per quadrat	30.42 ± 30.20		48.08 + 30.97
		1.74*[1]	
Mortality			
Living	95 (95)		269 (79)
Dead	13 (5)		72 (21)
		26.53***	
Sex			
Male	135 (50)		122 (45)
Female	137 (50)		147 (55)
		1.97NS	
Tussock diameter (cm)			
0-5	35 (10)		86 (15)
5-10	121 (33)		206 (36)
11-25	164 (45)		216 (37)
>25	48 (13)		70 (12)
		25.95***	

[1]Paired t-statistic

Values in brackets refer to percentages of individuals in categories.

* p = <0.05. ** p = <0.01. *** p = < 0.001. NS not significant

5. DISCUSSION

A managment programme which aims at the maximisation of sustained yield of a desirable species inevitably leads to a shift in relative abundance of coexisting species (Westman, 1990). This change in the community is not necessarily compatible with the requirements for the conservation of community diversity and maintenance of species richness (Cowling *et al.*, 1986).

The results from this study indicate that the effects of bushcutting on the *T. insignis* population are desirable for the increased production of thatching reed. The decrease in reproductive individuals following bushcutting may have important implications for the sustainable harvesting of the species. A study of the seed biology of the species in bushcut and non-bushcut sites would shed more light onto this aspect.
The promotion of the growth of *Cynodon dactylon* a palatable grass species in bushcut sites is also desirable as the veld can also be used for cattle grazing.

Disturbance is generally viewed as an event, intrinsic or extrinsic, which results in the removal of biomass or individuals that directly or indirectly create opportunities for the establishment of new individuals, which causes abrupt changes in the structure of a community (Sousa, 1984; Rykiel, 1985). Thus, the effects of disturbance can be seen at the population level. The community disturbance regime is defined in terms of the disturbance type, its size, duration, intensity, frequency and season of occurence. The commmunity disturbance regime is often the major selective force on life history strategies and community structure and functioning in general. In this study bushcutting can be seen as a disturbance.

The domination of disturbed sites by vegetation other than the previous dominants occurs in a variety of plant communities, especially after human disturbances. Cowling and Pierce (1988) studied succession after bushcutting in coastal dune fynbos in the South eastern Cape. They found that ten months after bushcutting, species richness was lower than in undistubed fynbos, yet after 1.4 year, richness increased two-fold. More than half the species present in the mature fynbos, including the dominants, had re-established 1.5 years after disturbance.

There is evidence from a number of sites in the Cape Floristic Region that understorey (and overall) species richness declines with increasing density and cover of the proteiod overstorey (Cowling and Gxaba, 1990). Despite this the non-bushcut vegetation (with a dominant proteiod cover) still had a greater number of species than the bushcut vegetation, of a comparable area. This indicates that bushcutting is an important determinant of species richness in mesic proteiod fynbos of the Riversdale coastal plain.

Recruitments after bushcutting to the flora of the stand of the size sampled here have interesting aspects. Some were probably

apparent gains rather than real, arising from the problem of inadequate sample size, i.e. not sampling all species representative of non-bushcut vegetation, due to time constraints.
Differential establishment of species after disturbance may result in changing dominance patterns (Bond, 1984). It appears that the bushcut community is open to colonization by species from species pools in the non-bushcut communities surrounding it. This is attributed to a `mass effect, i.e. the establishment of species in sites where they are not self-sustaining in the long term, as a result of the continual rain of propagules fom adjoining sites (Schmida and Wilson, 1985). Cowling (1990) suggests that for the Agulhas Plain, a mass effect is unlikely to affect alpha diversity, on a local scale, due to the low number of habitat generalists that can transgress community boundaries. In this study the habitat was fairly uniform so mass effect could effect species diversity. Studies are needed on the Riversdale coastal plain to assess habitat specificity of the flora, especially in view of future mass bushcutting in the area.

Dispersal ability differs enormously among plant species. Also, the greater the contrast between the non-bushcut vegetation and the bushcut vegetation the lower the species' dispersal rate between them (Diamond, 1975). In this study the community types were not sufficiently isolated from each other to influence colonization rates dramatically. This is not particularly surprising as many of the animals in the fynbos biome responsible for seed dispersal and pollination are generalists (Bond and

Slingsby, 1983; Rebelo, 1987). However, differential disruptions to the different dispersal distances and types was evident. In certain cases, the gains involved small scale-migrations. Long distance dispersal should easily occur.

Distance effects may become increasingly more important in other areas or become stronger as more natural vegetation is destroyed by bushcutting or for other agricultural uses. Serious disruptions to specialised pollination and dispersal mutualisms may occur. Species extinction as a result may be great.

One would expect that those species with obligate dependence on particular mutualists might be poorly represented in bushcut vegetation. In this study however, bushcutting did not drastically affect dispersal mutualisms, as seen through colonization failures. There was some evidence however for an absence of specialized pollinators in bushcut vegetation.

Some shrubs are able to resprout after bushcutting. This attribute enables populations of these species to recover after bushcutting. For example, the reprouting shrub *Erica tubular* (Ericaceae) would be better able to withstand bushcutting than the reseeders *Simocheilus dispar* (local endemic), *Anomalanthus scoparius* or *Platycalyx pumila* (Ericaceae) (personal communication - E.G.H. Oliver).
A predictive inderstanding of post-disturbance recruitment patterns requires studies on seedbank composition and dynamics, germination cues and mortality trends.

It is interesting to note that the only species in the bushcut samples contributing cover to height classes greater 50 cm was the dominant species *T. insignis*, which was probably intentionally not cut. The same species contributed all the cover in the graminoid growth form class (the largest class). This suggests that bushcutting eliminates all other previously occurring graminoides.

As can be seen by the presence of certain species groups surviving bushcutting, conditions after disturbance are suitable for the coexistence of short-lived species and longer lived fynbos shrubs generating from wind-borne and soil-stored seeds and resprouting species.

As a result of frequent bushcutting of the proteiod fynbos it is predicted that the vegetation can be converted to a *Thamnochortus insignis* dominated low grassy shrubland with a resprouting non-proteiod shrub cover, ericoids (*Passerina vulgaris*) and annuals (*Senecio* sp.).

The most interesting pattern is the under-representation of local endemic species in the bushcut vegetation, indicated by a decrease in the frequency and in the percentage cover of local endemics found in the bushcut vegetation compared to the non-bushcut vegetation. Bushcutting may thus reduce population sizes of local endemics to such an extent that it may lead to increased risk of extinction (Shaffer, 1981).

The composition and structure of the community changed across the farm-fence with an increase in patchiness. This is serious as the community which replaces it, is unstable, and prone to soil erosion, which may be increased with grazing pressure. It evident that the bushcut site was more heavily invaded by invasive shrubs such as *Acacia cyclops*.
Other secondary effects of bushcutting (for example fire and rodent activity) can modify the effects of bushcutting.

Plant response depends on the intensity, frequency and time of bushcutting with respect to the growing season, all of which are variable from farm to farm. The plant response to bushcutting may also be modified by site conditions such as substrate type. At present, mechanical bushcutting of proteiod fynbos is not possible on limestone outcrops. The results of this study cannot be used to predict vegetation change resulting from bushcutting in these habitats. Plant response (as shown in this study) is also dependent on the attributes of the species present. Thus, the results of this study are likely to be locality specific. They do however give an indication of what types of species are better able to survive bushcutting. Future studies are needed at a sufficient number of sites to evaluate the consistency of species response to bushcutting and to identify further species vulnerable to extinction as a direct result of bushcutting.

It is presumed that proteiod fynbos vegetation in productive stands of *T. insignis* would be repeatedly bushcut (approximately

ever five years - personal communication Mr C. Pretorius and Mr P. de Kok). Future studies need to examine the effects of repeated bushcutting on the proteiod fynbos.
Differences in population sizes (for example annual legumes) between years, as a result of rainfall distributions, for example, may effect the results of a once-off study (Noy-Meir *et al.*, 1989) such as the present study. The direction and significance of across fence differences in species abundance needs to be checked for consistency.

The Cape Floristic Region occupies 90 000 km^2 in the southern African Mediterranean climate zone (Bond and Goldblatt, 1984). The area has the highest recorded ratio of vascular plant species to area for any temperate and subtropical region (Rebelo and Siegfried, 1990) and is a well established centre of endemism (Cowling *et al.*, in press). The Bredasdorp-Riversdale centre, comprises a well-definded centre for the calcicole fynbos taxa confined to the Bredasdorp formation limestone and associated colluvial deposits which have their maximum exposure in this area (Rebelo and Cowling, 1991; Cowling and Holmes, in press). The occurence of three distinct vegetation communities, characterized by endemic Proteaceae taxa indicate that the Riversdale coastal plain is a centre of endemism, although not of the same order as the Agulhas plain. The coastal plain contains the greatest extent in the Cape region of many vegetation types (described by Rebelo *et al.* 1991).

In a study by Cowling and Holmes (in press) on the endemism of the larger families on the Agulhas plain the Ericaceae, Rutaceae (especially *Agathosma*), Proteaceae, Polygalaceae (especially *Muraltia*), Rhamnaceae (especially *Phylica*) and Mesembryanthemaceae were strongely over-represented in terms of both regional and local endemics. All these taxanomic groups were common in non-bushcut vegetation at the present study site. From the present study, it was evident that species from most of these families are vulnerable to extinction (medium-tall reseeding shrubs) as a result of bushcutting. Supposing these families are also over-represented in terms of endemics on the Riversdale plain [which has similar habitats and vegetation communities (Rebelo *et al.*, 1991; Cowling *et al.*, 1988)], the effects of future widespred bushcutting will serious. The same applies to woody plants, dwarf shrubs and species with short disperal distances, particularly ant dispersed species.

Many studies have indicated a correlation between endemism and a non-sprouting habit (Cowling *et al.*, 1991; Wells, 1969). If this association also holds for the vegetation of the Riversdale plain, the effects of future widespred bushcutting will be serious, as reseeding shrubs are under-represented in bushcut vegetation.

In a survey on the influence of agriculture on the decline of the west-coast renosterveld, MacDowell (1988) pointed out that land that at present appears safe from agricultural exploitation because of the steepness, lack of water, poor soil, etc., may not be necessarily safe because of agro-technical (and other) innovations. This may apply to the cultivation of *T. insignis* if in the future farmers are able to easily bushcut vegetation growing on rocky limestone outcrops and steeper slopes. This could result in the natural vegetation patches becoming 'islands' in a sea of agricultural land. According to the equilibrium theory of island biogeography, insularisation will lead to species loss from habitat remnants (MacArthur and Wilson, 1963). The aim of this study was to investigate the effects of bushcutting and to hopefully prevent such a disaster.

6. CONCLUSIONS

Qualitative and quantitative vegetation changes occur when *Leucadendron galpinii*-dominated Mesic Proteiod Fynbos communities are bushcut. Further research on the flora and vegetation of the Riversdale coastal plain is required before ecologists know the consequences of bushcutting vegetation in the whole region. This research has shown that bushcutting to increase thatch production has a pronounced effect on the biodiversity of the mesic proteiod fynbos communities and certain taxa are threatened with extinction as a result. Threatened taxa are often good indicators of threatened habitats, since they are much more sensitive to changes in the environment (Tansley, 1988a as cited by Hilton-Taylor and Le Roux, 1989) than common, widespred species. They can therefore be used in conservation planning in the establishment of conservation priorities.

It is suggested that the practice of bushcutting be confined and reserves set up immediately for the preservation of the vegetation of the Riversdale coastal plain. As pointed out by Rebelo and Siegfried (1990) the protection of Fynbos in the lowlands is more problematic compared to the montane areas, as centres of species richness are more diffuse, and endemic taxa tend not to show concordant distributions.

Most of the land on the Riversdale coastal plain is privately owned and not subject to state control. An alternate to the creation of state reserves is to entice the larger land owners to consider setting aside a portion of their land to be excluded from bushcutting. There are however problems with this suggestion as the conservation of threatened communities creates the problem of the withdrawal of high potential land from agriculture (Cowling, 1984; Lubke *et al.*, 1986). The Department of Nature and Environmental conservation have inadequate financial resources for the purchase of land of high agricultural potential (Cowling *et al.*, 1986).

Thus the communities in the Riversdale area are severely threatened because they occur on land that has the potential of being very lucrative.

The Riversdale coastal plain is clearly a phytogeographical unit which requires urgent conservation measures to ensure the preservation of its vegetation (Rebelo *et al.* 1991). The siting of reserves is very important. Ideally the localization of endemics and the extent to which a particular habitat type is threatened should provide a focus for reserves (Rebelo and Cowling, 1991; Rebelo and Siegfried, 1990).

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