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Changes in Arachnid Communities resulting from Forestry Practices in Karri Forest, South-west Western Australia

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Abstract

Arachnids were sampled by pitfall trapping in two areas of karri forest in the south-west of Western Australia on both ridge and creek sites. One area was sampled in spring and early summer for 4 years during which it was cleared, burnt and replanted with karri. The other site, sampled for 2 years, was mature karri forest. Five orders of Arachnida, including 18 families of Araneae, were represented. Clearing and burning resulted in more aggregated captures and in marked changes in the structures of the arachnid community. There were changes in species dominance and a proportional increase in the number of individuals at creek sites. Species richness recovered from the forestry management practices more slowly on the creek sites than on the ridge sites.

[O.D.C. 145.4 : 178.6 *Eucalyptus diversicolor*]

Introduction

Evidence is accumulating that, collectively, spiders promote the stability of arthropod populations (see e.g. Moulder and Reichle 1972; Riechert 1974; Enders 1975). Accordingly, they warrant consideration in forest management practices aimed at maintaining a persistent ecosystem. Spider communities are influenced considerably by forest management practices (Huhta 1971; Coyle 1981) and by the successional stages in the development of communities (Merrett 1976; Bultman *et al.* 1982). A general increase in species diversity has been observed through early and mid successional stages, followed by a decrease in diversity in the climax community (Huhta 1971; Bultman *et al.* 1982).

Ecological data on the ground-dwelling invertebrates in Western Australia are sparse, although in recent years their seasonal and post-fire activities have been studied by Springett (1976*a*, 1976*b*), Koch and Majer (1980) and Majer and Koch (1982). Mawson (1983) reported on their recolonization after bauxite mining and rehabilitation. Here we record the activity of arachnids in karri forest during spring and early summer in 4 consecutive years during which the area was cleared, burnt and replanted with karri.

Study Site

The study area was in the Crowea forest block, 12 km SE. of Pemberton (34° 28′ S., 116° 10′ E.) and had an estimated rainfall of 1400 mm. The upper slopes had jarrah/marri (*Eucalyptus marginata/E. calophylla*) forest on podsolic soils, while in the valleys, where trapping was conducted, karri (*E. diversicolor*) forest grew in red earths. Within this area the understorey near creek sites was *Oxylobium lanceolatum* and *Trymalium spathulatum* with a thick ground cover of

Lepidosperma tetraquestrum. Ridge sites had an understorey of *Allocasuarina decussata* and three *Acacia* species: *A. urophylla*, *A. browniana*, *A. divergens*.

In 1976, study area A consisted of closed karri forest. Between January and July 1977 the area was clear felled and all commercial timber removed. In April 1978 the standing vegetation and fallen timber were burnt and the area was replanted in July 1978. By 1979 the karri trees were up to 3 m high and there was a dense shrub layer. Study area B comprised closed forest in both years of sampling (Table 1).

Climatic data for the area are shown in Table 2, from which it should be noted that the first year was unseasonally cold during the sampling period.

Table 1. Sampling times and descriptions of sites for arachnid sampling

The sites were in karri forest at Crowea Coupe, south-west Western Australia

Area	Sampling period	No. of trap–weeks	Site description
A	16.x.76–24.xii.76	100	Uncleared closed karri forest
A	24.x.77–19.xii.77	160	Clear felled
A	25.x.78–20.xii.78	160	Burnt (iv.78), replanted (vii.78)
B	25.x.78–20.xii.78	160	Closed karri forest
A	25.x.79–20.xii.79	160	Karri regrowth to 3 m, dense herbs and shrubs
B	25.x.79–20.xii.79	160	Closed karri forest

Table 2. Climatic data for study area

The data are from the Pemberton Forestry Station (34° 27′ S., 116° 03′ E.; alt. 174 m), Bureau of Meteorology. The means are: S_x, November–December mean for the years in which trapping was conducted for arachnids in karri forest; A_x, annual mean for each year 1976–79; C_x, long-term (24–33 years) climatic mean

Year	Mean	Temperature (°C) min.	Temperature (°C) max.	Relative humidity (%)	Rainfall (mm)
1976	S_x	10·7	21·4	64	134
	A_x	10·6	20·7	70	1124
1977	S_x	12·6	24·0	61	72
	A_x	10·3	20·7	69	1008
1978	S_x	12·0	24·4	55	86
	A_x	10·4	25·4	—	1166
1979	S_x	11·5	23·3	66	142
	A_x	12·3	—	—	—
Long-term	C_x	9·9	19·9	68	1245

Methods

The two study areas A and B were located within 2 km of each other, and at each site sampling was conducted beside a creek and on a ridge about 240 m distant. The site descriptions and sampling times are shown in Table 1. The pitfall traps consisted of 200-mm glass test tubes with 40 mm internal diameter and were located within PVC sleeves to avoid disturbance when sampling (Majer 1978). Adjoining tubes were placed 1 m apart and contained a mixture of ethanol, water and glycerol (56:24:20 v/v). The tubes were left open for each 7-day sampling period of 10 weeks in 1976 and 8 weeks in 1977, 1978 and 1979 (Table 1).

Specimens were pooled from each site at each sampling period and lodged with the Western Australian Museum.

Two indices of association were used: the Jacaard index for the presence/absence data and the Bray and Curtis index for the numeric data (Southwood 1978). The indices were clustered by means of the WPGMA method (Sneath and Sokal 1973). The statistical significance of the clusters was determined using the procedure of Strauss (1982). This involved randomizing the original data set by holding the number of species constant and allocating those species present a random number between 1 and the maximum number of that species in the original data set. The random matrix was then clustered as above and the dissimilarity value accumulated for each node level. The process was repeated until more than 3500 nodes were accumulated and the 95% limits were calculated from the distribution of each node level. Additionally, an index of information content (H') was calculated (Shannon 1948); this has been used elsewhere in studies on cursorial spiders (Uetz 1975, 1976, 1979; Uetz *et al.* 1979; Bultman *et al.* 1982) and is a good index for pitfall samples (Pielou 1966). The estimated number of taxa for a given number of individuals was estimated by rarefraction (Sanders 1968) using the corrected procedure and algorithms of Heck *et al.* (1975) and Simberloff (1978).

Results

A total of 916 individuals in 55 taxonomic categories were trapped on the sites (Table 3*a*). Five orders of Arachnida were present with the Araneae (spiders) represented by 18 families. The diversity (H') and evenness (J') indices are given in Table 3*b* for the individual sites and for the combined sites in each year.

Inspection of Table 3 shows that the two sampling areas are not directly comparable so that area B cannot be used as a pre-treatment control for area A. For example, the potential control area B has a statistical excess of scorpions, *Storena variegata* and *S. scintillans*, while it has a deficit of phalangids (all by χ^2; $P<0{\cdot}05$).

Clustering the site data by year (Fig. 1*a*) suggests that there were large differences between areas A and B and that the differences between ridge and creek sites were small compared with those between years and areas. Moreover, the effect of clearing *per se* had little short-term effect on the similarity indices of the community considered, but burning had profound effects and resulted in statistical discontinuity. This is more clearly shown by considering area A alone (Fig. 1*b*).

In area A the number of species before and after burning did not alter significantly but the distribution of individuals changed markedly (contingency χ^2 with 1 d.f. $= 47{\cdot}31$; $P<0{\cdot}001$). (Before burning 68% of individuals were trapped on ridge sites, whereas after burning 63% were caught at creek sites.) The same change, however, occurred between years (57% and 61% respectively) in area B (contingency χ^2 with 1 d.f. $= 13{\cdot}10$; $P<0{\cdot}001$).

The H' diversity was similarly high in area A before clearing and in the untreated area B over 2 years, being respectively 3·88, 3·85 and 4·34 (Table 3*b*). This measure of diversity decreased by 31% after clearing, before gradually increasing to reach 89% of the pre-cleared value in the first year of regrowth. Creek sites were always less diverse than ridge sites in area A, but the converse prevailed in area B. For the karri forest, the recorded diversity was highest in the mature forest and lowest after clearing.

Table 3. Data by site and year for species of arachnids pitfall-trapped in karri forest in south-west Western Australia

In (*a*) the site treatments are: U, uncleared; C, cleared; F, burnt; R, regrowth. In (*b*) are given the total number N of individuals, the total number S of species (identified taxa), the Shannon (1948) diversity index H' (log base 2) and the evenness index J' ($= H'/H_{max}$) for each area (A and B) and year (1976–79), and for ridge and creek sites combined

(*a*) *Individual data*

No.	Species–taxon	Area A								Area B			
		Ridge site year, treatment				Creek site year, treatment				Ridge site year, treatment		Creek site year, treatment	
		1976, U	1977, C	1978, F	1979, R	1976, U	1977, C	1978, F	1979, R	1978, U	1979, U	1978, U	1979, U
1	**Acarina**	2	1	0	0	0	0	0	0	0	0	1	8
	Scorpionida												
	Bothriuridae												
2	*Cercophonius squama* (Gervais, 1844)	2	0	4	0	1	1	1	0	9	5	10	3
3	**Pseudoscorpionida**	0	0	0	0	0	0	1	0	1	0	1	0
4	**Phalangida**	6	17	3	0	5	4	2	4	4	0	0	3
	Araneae												
	Mygalomorphae												
	Dipluridae												
5	*Chenistonia villosa* Rainbow & Pulleine, 1918	14	27	1	1	14	18	0	1	15	14	6	3
	Araneomorphae												
	?Amaurobiidae												
6	*Baiami brockmani* Gray, 1981	9	3	4	9	4	0	5	7	25	1	1	0
7	Genus A, sp. A	1	1	0	0	0	0	0	0	0	1	1	0
8	*Badumna* sp.	1	0	0	0	0	0	0	0	0	0	0	0
	Salticidae												
9	Genus A, sp. A	3	1	0	0	2	1	0	2	1	5	0	5
10	Genus A, sp. B	1	0	0	0	1	0	0	0	0	2	0	0
11	Genus B, sp. A	1	1	0	0	0	0	0	0	0	1	0	0
12	Genus C, sp. A	0	0	0	1	0	0	0	0	0	0	0	0
13	*Saitis* sp.	0	0	0	1	0	0	0	0	0	0	0	0
	Clubionidae												
14	*Myandra* sp.	1	1	0	0	0	0	0	0	0	1	4	3
15	*Supunna michaelseni* Simon, 1909	0	0	0	1	0	0	1	0	0	2	0	0
16	?*Supunna* sp. A	0	0	1	1	1	0	1	0	1	3	0	0
17	Genus A, sp. A	0	0	0	0	0	0	0	0	0	1	1	4

	Miturgidae												
18	*Miturga* sp.	0	0	0	0	0	0	0	0	0	0	0	1
19	Genus A, sp. A	0	0	0	0	0	0	0	0	4	2	0	0
	Gnaphosidae												
20	Genus A, sp. A	0	3	0	0	0	0	0	0	0	3	14	1
21	Genus B, sp. A	0	1	0	1	0	0	0	0	2	2	3	5
22	Genus C, sp. A	0	0	0	3	0	0	0	5	0	0	1	1
23	Genus D, sp. A	0	0	0	0	1	0	0	1	2	0	1	1
24	Genus E, sp. A	0	0	0	0	0	0	0	0	1	0	1	2
25	Genus G, sp. B	0	0	0	0	0	0	0	0	1	0	0	1
26	Genus H, sp. A	0	0	0	0	0	0	0	0	1	0	0	1
27	Unidentified	0	0	0	1	0	0	0	0	0	0	0	2
	Thomisidae												
28	*Sidymella* sp.	0	1	0	3	0	0	0	1	0	1	4	0
29	Genus A, sp. A	0	0	0	0	0	0	0	0	0	0	0	1
	Ctenidae												
30	*Elassoctenus* sp. A	0	2	0	0	2	2	0	0	1	3	0	1
31	*Elassoctenus* sp. B	1	1	0	0	1	0	0	0	12	1	0	1
32	Genus C, sp. A	1	0	0	0	0	0	0	1	0	0	0	0
33	Genus D, sp. A	0	0	0	0	0	0	0	0	0	0	0	2
	Theridiidae												
34	Genus C, sp. A	0	0	0	0	0	0	2	0	0	0	0	0
35	*Steatoda* sp.	0	1	3	2	2	0	2	1	0	1	0	1
36	*Nicodamus* sp.	0	0	0	0	0	0	0	0	0	0	0	1
37	Genus B, sp. A	0	0	0	3	0	0	4	4	0	0	0	0
38	Unidentified	0	0	0	0	0	0	3	1	0	0	0	0
	Zodariidae												
39	?*Storena variegata* Cambridge, 1869	0	0	0	1	1	4	0	0	6	10	16	24
40	?*Storena scintillans* Cambridge, 1869	0	0	0	1	0	0	0	0	27	0	16	8
41	*Storena tetrica* Simon, 1908	0	0	0	0	0	0	0	0	1	3	3	2
	?Agelenidae (or Desidae)												
42	Genus A, sp. A	0	7	0	0	0	0	0	3	2	7	1	2
43	Genus B, sp. A	0	0	4	0	0	0	0	0	0	0	0	0
	Lycosidae												
44	*Artoria cingulipes* Simon, 1909	16	19	3	6	13	1	10	0	0	3	0	2
45	Genus B, sp. A	10	1	1	2	1	0	1	0	1	2	0	0
46	Genus B, sp. B	1	0	0	1	0	0	0	0	0	2	0	0
47	Genus C, sp. A	2	0	4	3	0	0	13	0	0	0	0	0
48	Unidentified	2	1	13	1	1	0	20	46	6	2	7	31

Table 3 (*Continued*)

No.	Species–taxon	Area A								Area B			
		Ridge site year, treatment				Creek site year, treatment				Ridge site year, treatment		Creek site year, treatment	
		1976, U	1977, C	1978, F	1979, R	1976, U	1977, C	1978, F	1979, R	1978, U	1979, U	1978, U	1979, U
	Araneomorphae												
	Orsolobidae												
49	*Tasmanoonops* sp.	2	0	0	0	4	0	0	0	0	0	0	1
50	Micropholcommatidae	3	0	0	0	0	0	0	0	0	0	0	0
51	Araneidae	3	0	0	0	0	0	0	0	0	0	0	0
52	Pisauridae	11	0	0	0	0	0	0	1	0	0	0	0
53	Linyphiidae	1	0	1	0	1	0	0	0	0	0	0	0
54	*Ostearius melanopygius* (Cambridge, 1879)	0	0	3	0	0	0	0	0	0	0	0	0
	Hahniidae												
55	Genus A, sp. A	0	0	0	0	0	0	0	2	0	0	0	0

(*b*) *Aggregate numbers and indices*

Grouping of ridge and creek sites	Number or index	Area A								Area B			
		Ridge site year, treatment				Creek site year, treatment				Ridge site year, treatment		Creek site year, treatment	
		1976, U	1977, C	1978, F	1979, R	1976, U	1977, C	1978, F	1979, R	1978, U	1979, U	1978, U	1979, U
Separate	*N*	94	89	45	42	55	31	66	80	123	78	92	121
	S	23	18	13	19	17	7	14	15	21	25	19	29
	H'	3·82	2·99	3·29	3·80	3·33	1·95	3·04	2·44	3·46	4·14	3·52	3·79
	J	0·84	0·72	0·89	0·89	0·81	0·69	0·79	0·62	0·79	0·89	0·83	0·78
Combined	*S*	28	20	18	27					28	28		
	H'	3·88	2·96	3·38	3·46					3·85	4·34		
	J	0·80	0·68	0·81	0·73					0·80	0·82		

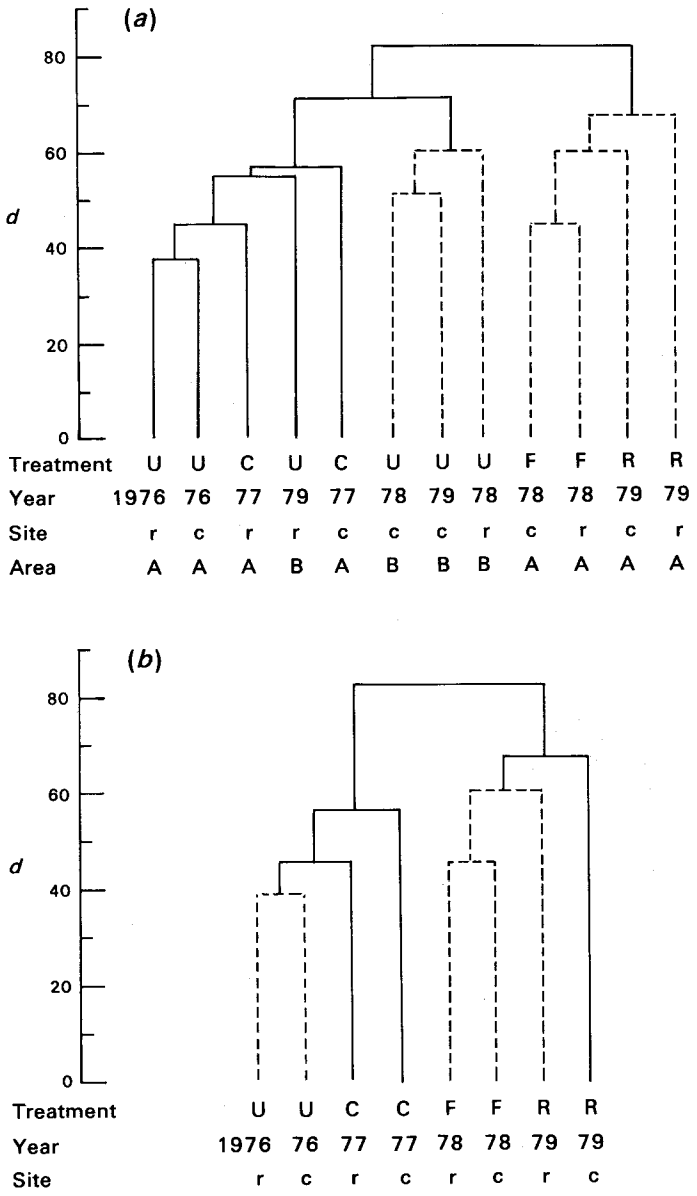

Fig. 1. Dendrograms of dissimilarity (*d*) values based on numeric data clustered by means of the WPGMA method, for (*a*) all years and sampling sites in the karri forest and (*b*) area A only. Solid lines denote significant linkages at a 95% probability level, determined as described in the text. The treatments at the ridge (r) and creek (c) sites were: U, uncleared (natural) forest; C, cleared; F, burnt; R, replanted.

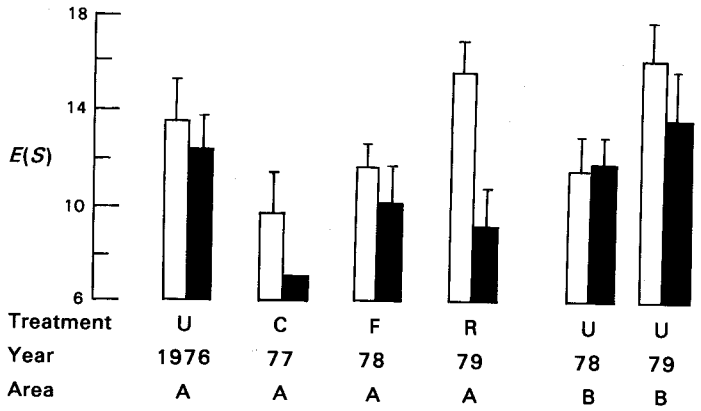

Fig. 2. Numbers of taxa per 30 individuals, $E(S)$, calculated by rarefraction. Open columns represent the ridge sites, and solid columns the creek sites; the vertical bars indicate a standard deviation. Other notation is as for Fig. 1.

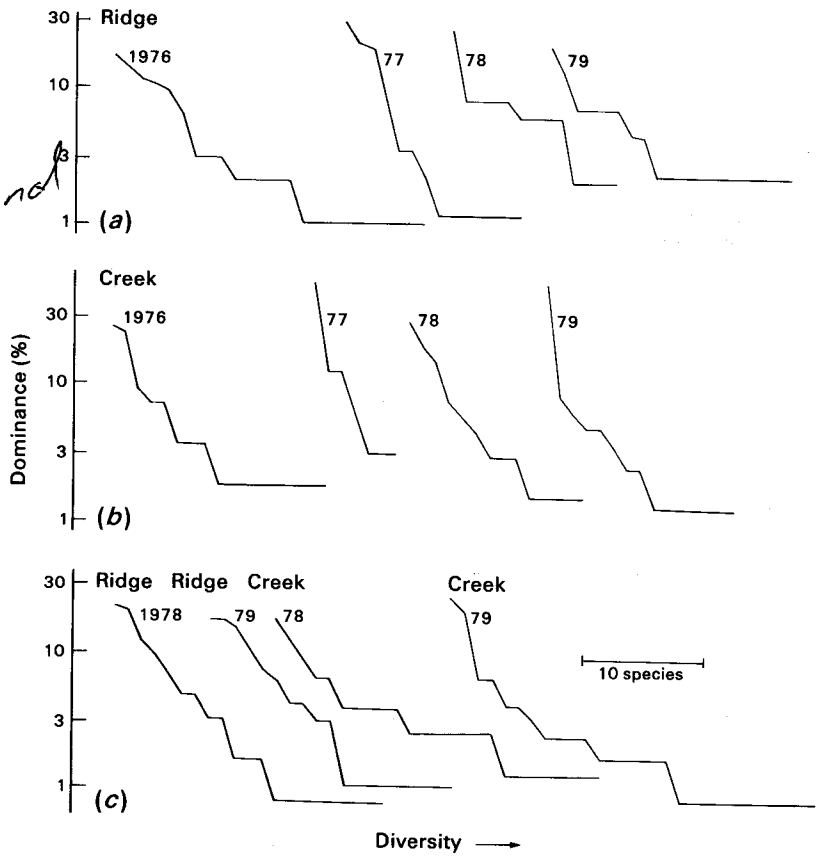

Fig. 3. Dominance–diversity curves for the arachnid communities sampled in the karri forest: (*a*) area A, ridge sites; (*b*) area A, creek sites; (*c*) area B, all sites.

Comparison of species richness between creek and ridge sites was examined further using the estimated species richness calculated by rarefraction, to eliminate the effect of sample size (Fig. 2). Although there was considerable between-year variation in area B, within years the ridge and creek sites were similar in uncleared forest. In area A, species richness remained low on the creek sites after clearing, while it increased substantially on the ridge sites after replanting.

No significant difference occurred in the number of taxa between sites, between habitats, or before or after burning. Large changes did occur, however, and they are clearly shown by the dominance–diversity curves in Fig. 3. These indicate, by their change in slope and tail length, that the community was less species-rich following clearing and that this effect lasted into the regrowth sample.

Individual taxa were differentially affected by treatment (before *v.* after burning). Diplurids were more abundant on ridges than on creeks after burning compared with before burning (contingency χ^2 with 1 d.f. = 4·47; $P<0·05$); they were also more abundant on ridges in area B (contingency χ^2 with 1 d.f. = 5·26; $P<0·025$). The most notable changes occurred in the lycosids: relatively more of the identified lycosids were trapped on the creek sites after burning (contingency χ^2 with 1 d.f. = 10·91; $P<0·001$), and on both ridge and creek sites there was a qualitative change with more of the unidentified lycosids occurring after burning (contingency χ^2 with 1 d.f. = 16·1 and 47·3; $P<0·001$ and $<0·001$ respectively).

Table 4. Negative binomial statistics describing dispersion of trapping data

All data sets (with ridges and creeks combined for each year) gave acceptable fits to negative binomial distributions fitted by the maximum likelihood method. The values of k give a measure of the dispersion and P is the probability of the distribution not being a negative binomial

Area	Year	Mean	k	P
A	1976	1·38	0·23	0·35
	1977	1·11	0·10	0·14
	1978	1·03	0·13	0·18
	1979	1·13	0·18	0·41
B	1978/9	1·91	0·25	0·57

To examine the effects of the management procedures on the dispersion of the trapping data, the latter were fitted to theoretical distributions. All data sets were significantly 'overdispersed' (clumped) but gave acceptable fits to the negative binomial distribution fitted by the maximum likelihood method (Table 4). All the data show extreme clumping but the undisturbed forests in both areas A and B show similar dispersion (k) values. Following clearing, the trapping data show greater clumping (in 1977), after which the k values rise towards, but do not reach, the undisturbed levels. This change in community structure is most clearly seen in the dominance–diversity curves (Fig. 3).

Discussion

Six pitfall traps (15 cm diam. and spaced 6 m apart) have been shown to sample cursorial spiders adequately and to give a reliable estimate of the number of species over a wide range of habitats (Uetz and Unzicker 1976). They do not permit a true

estimate of density but sample the number moving in the area during the sampling period—Uetz's (1977) 'active density'. Before further discussion, some consideration is required of the likely effects of unequal sampling effort in the first 2 years (see Table 3).

Sampling intensity in the first 2 years was respectively 63% and 88% of the final 2 years of sampling. This does not affect analyses examining the distribution of arachnids between ridge and creek sites. It does, however, mean that only a decrease in numbers between 1976 and the other years and between 1976–77 and the other years can be examined, since the numbers would be expected to increase owing to different sampling intensities. Several effects of the management procedures on the arachnid community can be putatively identified from the data but a rigorous sampling design would be required to confirm them.

The structure of the arachnid community changed in response to the treatments and the subsequent successional changes. Species diversity H' was high prior to clearing, decreased after clearing and increased with subsequent regrowth. This trend was accompanied by changes in the distribution of arachnids which were more clumped after clearing and subsequent treatments. The clumping may be associated with the burnt timber and its associated ashbeds; the latter would be expected to cause long-term patchiness in the area.

Diversity in karri forest was highest in mature forest and lowest after clearing. If the karri forest community behaves consistently with those in other areas studied (Huhta 1971; Bultman *et al.* 1982) we may expect that continued regrowth would result in an increase in diversity above the levels found in the mature forest, followed by a gradual decline.

With one exception, the measure of evenness (J') was greater for ridge sites than for creek sites in both areas A and B. Evenness decreased markedly following clearing on ridges and creeks but subsequently increased on the ridges while depressed on the creeks in 1979. Both evenness and H' diversity changed concordantly on both ridge and creek sites in area B. The rarefraction analysis (Fig. 2) supports this differential response between ridge and creek sites to clearing and subsequent treatment. The creek sites showed a much slower recovery in their species richness compared with the ridge sites.

Considering only spiders in area A for the last 3 years in which sampling intensity was similar, there was a decrease in the number of individuals caught after burning on the ridges ($P<0{\cdot}05$) and an increase in the number trapped on the creek sites ($P<0{\cdot}001$). This consistent change in the distribution of spiders in area A after burning contrasts with area B, where the between-site distribution varied between years (contingency χ^2 with 1 d.f. = 10.81; $P<0{\cdot}005$). After the burns, fewer individuals were trapped on the ridge sites than on the creek sites. This suggests that the ridges provide important refuges for the spiders. Lycosids were more abundant in the plots after burning (reset succession) and this is consistent with them being more abundant in early successional stages in other areas (Bultman *et al.* 1982).

Examination of different guilds of arachnids indicates differential effects on them of the different procedures adopted. The lycosids ('sit-and-wait' predators with good dispersal ability) were affected immediately by clearing but returned, albeit with different species, immediately after burning. The pisaurids disappeared after clearing and returned only after replanting. The diplurid, *Chenistonia villosa* (also a 'sit-and-wait' predator but confined to a burrow and hence having poor

power of dispersal), was not influenced by clearing but was almost eliminated by burning. Ridge sites probably became drier following the removal of vegetation, but we have too little information on the characteristics of the site and on individual species to attempt an analysis of this factor.

The effects reported here of forestry practices on arachnid communities are small in comparison with those effects on areas in which native forest has been replaced by conifer plantations. In the Otway Ranges, Victoria, mature *Pinus radiata* plantations were found to support fewer species (38%) and smaller populations (36%) than adjacent mature *Eucalyptus regnans* forest (Ahern and Yen 1977). At Gnangara in Western Australia, mixed *Eucalyptus*, *Banksia* and *Hakea* woodland was more diverse than 31-year-old *Pinus pinaster* plantations, with 80% more species overall and 133% more predatory species (Springett 1976*b*).

Springett (1976*a*) also compared, for both jarrah and karri forests, the arthropods in areas of long-unburnt forest with those in forest with a history of management by prescribed burning. Burnt jarrah forest had 22% and 50% of the individuals and species respectively present in unburnt areas. For karri forest the values were 31% and 81% respectively.

The work reported here examines short-term effects of forestry management procedures. Overall there was no prolonged reduction in species richness but there were considerable changes to the composition and structure of the arachnid community. Furthermore, there were considerable small-scale differences in the response to management practices, with creek sites showing longer term reduction in species richness than adjacent ridge sites. It follows that considerable stratification of sampling regimes may be necessary in order to gain an understanding of the response of arthropod communities to forestry management. Finally, it should be stressed that the recovery of species richness does not indicate that management procedures are not detrimental to the maintenance of natural communities — species richness has to be considered in conjunction with the composition and structure of the community.

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