SCOPE ITEM 7

IMPACTS OF DIEBACK-INDUCED VEGETATION CHANGES ON NATIVE FAUNAL COMMUNITIES IN SOUTH-WEST WESTERN AUSTRALIA

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1 INTRODUCTION

In February 1998, the Department of Conservation and Land Management (CALM) commissioned Environmental Management and Research Consultants (represented here by the writer) to review impacts of dieback-induced vegetation changes on fauna. The specific objective of the work was to review existing knowledge on the impact of *Phytophthora cinnamomi*, and of management practices for its control, on the welfare of native animal species and communities with a view to identifying opportunities and constraints for the abatement of those impacts.

In the course of the review it was found that little detailed research had been carried out in the subject area. Apart from work on mammals in Victoria, most studies had been conducted in Western Australia. Accordingly, the review concentrates mainly on the situation in the latter State and, to a lesser degree, in Victoria. Existing knowledge is summarised, results are synthesised and interpreted, and from the conclusions, recommendations are developed for the minimisation of dieback impacts and the conservation of potentially threatened species.

2 DIEBACK-INDUCED VEGETATION CHANGES

P. cinnamomi is a phytopathogenic fungus that causes serious root-rot and dieback of many Australian native plant species. The disease has significantly damaged various plant communities including tropical rainforests in Queensland, eucalypt forests, woodlands and heathlands in Victoria and the jarrah forest and heathlands of south Western Australia. The pathogen spreads autonomously by movement of water-borne zoospores in the soil and mycelial growth in infected plant roots, or through vectored spread of infested soil by man and other animals.

The distribution and severity of infection in plant communities is influenced by temperature, soil type, nutrient status and water availability (Weste & Marks, 1987; Marks & Smith, 1991; Wilson *et al.*, 1994), thus the magnitude of disease impact varies between sites. Greatest impact generally occurs where soils are infertile and drainage is poor. Communities growing on fertile soils, high in organic matter, may suffer relatively little damage (Shearer & Tippett, 1989).

Eucalyptus marginata (jarrah) is the dominant tree species in forests of south Western Australia and it is the only eucalypt in the jarrah forest that is susceptible to *P. cinnamomi*. Healthy jarrah forest has a dense understorey and shrub layer which is floristically rich and includes a diverse flora with nectar-producing flowers that are pollinated by small mammals, birds or insects. Many of these plant species, including *Xanthorrhoea preissii* and members of the Proteaceae, Dilleniaceae, Papilionaceae (Podger, 1968) and Epacridaceae (Keighery, 1988) are particularly susceptible to dieback. *Banksia grandis* is also highly susceptible and mortality of this species is one of the first indicators of infestation by *P. cinnamomi* (Shearer & Tippett 1989).

In the Proteaceae-dominated heathlands along the south coast of Western Australia, the impact of dieback is particularly severe, resulting in marked structural changes and decline in floristic diversity.

The effect of dieback infection on plant communities is similar in Victoria, although different species are present there. The cover provided by tree crowns decreases and the composition of the understorey is altered. Susceptible species such as *Xanthorrhoea australis* disappear or are greatly reduced in abundance while resistant species, particularly grasses and sedges, take over. In some situations, severe erosion occurs as a result of vegetation losses (Kennedy & Weste, 1986).

Many parts of the jarrah forest were severely affected by dieback between the 1950's and the 1980's. Frequently, all or most of the jarrah and much of the understorey was killed and replaced by other plant species. Resistant marri (*Eucalyptus calophylla*) recolonised affected sites which eventually became open woodland with a ground cover dominated by sedges (Shearer & Tippet, 1989). By comparison with the original forest, floristically impoverished communities developed in which the lower stratum consisted predominantly of plants that lacked nectar-producing flowers.

3 SUMMARY OF EXISTING KNOWLEDGE

3.1 MAMMALS

No specific studies on the impacts of dieback-induced vegetation changes on mammals have been carried out in Western Australia. Several of the Alcoa studies discussed in sections 3.2 and 3.3 have included mammal trapping, but for most species the numbers trapped were too low to detect any differences that might have existed.

Friend (1992) hypothesised that the Honey Possum (*Tarsipes rostratus*) could become a species under threat as a result of dieback impacts on floral communities, particularly in south coast heathlands. Since many of the plant species on which the Honey Possum feeds are susceptible to *P. cinnamomi*, dieback infestation could significantly deplete the animal's food resource.

Wilson *et al.* (1994) speculated that habitat changes caused by *P. cinnamomi* and *Diplodina* (a cause of canker disease) could potentially threaten the Dibbler (*Parantechinus apicalis*) in south-west Western Australia by decreasing the number of prey invertebrates inhabiting flowers and the deep litter layer. They also concluded that *Pseudomys shortridgei*, a restricted mammal species which depends on floristically rich vegetation, could be detrimentally affected by the spread of dieback in the Victorian Grampians.

Specific research on the impacts of dieback-induced vegetation changes on mammalian fauna appears to have been conducted only in Victoria. Laidlaw & Wilson (1994) studied small mammals in open forest, woodland and heathland in Angahook-Lorne State Park, near Anglesea where some areas were affected by dieback. The study did not find any relationship between dieback impact and the diversity or abundance of small mammals. However, the authors noted the probability that insufficient sites were examined to allow detection of significant differences.

Wilson *et al.* (1994) cite an unpublished study by Laidlaw & Wilson as finding lower abundances of several small mammal species such as the Swamp Rat (*Rattus lutreolus*), Bush Rat (*R.. fuscipes*) and the Brown Antechinus (*Antechinus stuartii*) in dieback-affected heathland near Anglesea in Victoria. Mean species richness of small mammal communities was also found to be lower at infected sites.

Detailed studies were also carried out by Newell & Wilson (1993) who investigated the relationship between dieback, vegetation changes and the abundance of *A. stuartii* in Victoria's Brisbane Ranges. Trapping indicated that the species was significantly less abundant in sites infested with *P. cinnamomi* than in uninfested sites. The volume of vegetation at the 0-20 and 20-24cm heights was also significantly reduced. When the abundance of *A. stuartii* was regressed against vegetation, a positive significant correlation was found at both of the above heights. These results and other observations clearly indicated a link between dieback impacts, structural decline of vegetation and numbers of *A. stuartii*. Radio tracking studies by Newell (1994) indicated that *A. stuartii* showed a high degree of overlap with areas that were unaffected by dieback while the species largely avoided affected areas.

Newell & Wilson (1993) investigated the relationship further and concluded that the decline of the grass tree (*X. australis*) was a major contributing factor in the reduction of *A. stuartii* abundance, possibly because of the decrease in shelter associated with loss of the tree. Other studies conducted by Newell (1997) did not indicate significant declines in the abundance of ground-dwelling invertebrates, so reduced food availability would appear to be an unlikely reason for the decreased abundance of *A. stuartii*.

Wilson *et al.* (1994) provided a table of predicted effects on fauna associated with the presence of *P. cinnamomi* in plant communities. This is shown in Table 1 where it can be seen that widespread changes in the vegetation are possible with consequent effects on fauna. The predicted effects on fauna are discussed in more detail in Section 4.

3.2 BIRDS

Very little research has been published on the effects of dieback-induced vegetation changes on avifauna. Several research projects, conducted in the jarrah forest by Alcoa of Australia Ltd., have focused on comparisons between avifaunal recolonisation of rehabilitated bauxite minesites and bird communities in unmined forest. Unmined control plots included areas of healthy and dieback-affected forest, and were intended to represent a cross-section of pre-mining vegetation types. Useful comparative data on the avifauna of healthy and diseased forest were obtained in the course of monitoring. The following results were extracted from published work by Nichols & Watkins (1984) and from unpublished Alcoa data and reports.

Table 1. Predicted effects on flora and fauna due to the presence of *P. cinnamomi* in plant communities in Victoria (from Wilson *et al.*, 1994)

Effects on Vegetation	Effects on Fauna
1. Loss of susceptible plant species in the understorey or midstorey.	• Direct loss of food sources, *e.g.*, seeds, pollen. • Indirect loss of food sources, *e.g.*, invertebrates.
2. Decline in species richness and diversity.	• Loss of food for species that prefer floristically rich vegetation. • Loss of seasonal food availability.
3. Decrease in plant cover; increase in bare ground and erosion.	• Loss of habitat for species dependent on thick ground cover. • Increased predation risk. • Changes to microclimate.
4. Decrease in canopy cover.	• Loss of food for arboreal species. • Loss of habitat for arboreal species.
5. Decrease in litter fall.	• Decline in litter invertebrates (dry conditions). • Decline in invertebrate food sources for insectivores.
6. Post-infection increase in frequency of field resistant plant species, *e.g.*, sedges.	• Increase in food for specialist herbivores.

In 1981 a long term monitoring program commenced at Alcoa's Jarrahdale mine. Birds were surveyed within 20m of transects located in two healthy and two dieback-affected areas of forest. Methods are described by Nichols & Watkins (1984). The surveys were repeated in 1987 and 1993. In all years, monitoring was undertaken in summer. Impacts in both dieback-affected sites were severe, although some rehabilitation (understorey establishment and planting of resistant trees) was carried out in Dieback Site 1 in 1994. Results are summarised in Table 2 and complete data for each year is given in Appendix 1. Although insufficient sites were monitored to allow statistical analyses of results, the data suggest that several differences exist between the bird communities in healthy and dieback-affected forest.

Table 2. Numbers of bird species, densities (number/ha) and Shannon-Wiener Diversity (relative abundance) recorded in surveys of healthy and dieback-affected, unmined forest at Jarrahdale in 1981, 1987 and 1993 (results from Armstrong & Nichols, in prep.)

	Healthy 1	Healthy 2	Dieback 1	Dieback 2
No. of species				
1981	16	16	9	14
1987	18	20	13	10
1993	27	28	21	16
Total density				
1981	13.2	12.6	6	13.5
1987	9.5	13.2	6.2	6.2
1993	12.2	8	8.2	8
Diversity				
1981	1.09	1.08	0.88	0.99
1987	1.13	1.18	1.06	0.89
1993	1.03	1.15	1.04	1.04

The results summarised in Table 2 indicate that total numbers of bird species were consistently less in sites affected by dieback than in healthy forest. Bird density in dieback sites was also lower for some but not all surveys. In most cases, diversity (which compares the relative abundance of species) was reduced in the dieback-affected sites. These results indicate that dieback-induced vegetation changes have some affect on avifaunal communities.

The data in Appendix 1 give some indication of what those changes might be. The White-naped Honeyeater was recorded in healthy forest on most occasions but never in the dieback-affected sites. The Western Spinebill was usually recorded in healthy forest sites and in dieback Site 2, but not in Site 1. No honeyeaters were recorded at dieback Site 1 during any surveys. Since forest-dwelling honeyeaters commonly feed on susceptible plant species such as *B. grandis*, *Adenanthos barbigerus* and *Grevillea spp.*, these results suggest a link between dieback impacts and a local decline in honeyeaters.

Indirect effects on bird species are also possible. Many birds are insectivorous, thus any decline in insect numbers due to vegetation changes would be expected to affect the numbers of species that feed on insects. In Appendix 1 it can be seen that the

Rufous Treecreeper, a bird that eats insects on tree trunks, was usually recorded in healthy forest but never in diseased sites. The Western Yellow Robin, which forages for insects in the middle and lower strata, and the Grey Shrike-thrush, a mid-stratum forager, also tended to be more common in healthy than in dieback-affected areas. However, numbers of other insectivores such as the Striated Pardalote, a leaf gleaning species, and the Grey Fantail were similar in both healthy and dieback-affected forest. These results indicate that numbers of some insectivorous bird species may decline in severely diseased areas, but others are not so obviously affected. Further studies are needed to clarify the situation.

Structural changes in the vegetation may also have a negative effect on some bird species. However, the relationship is not clear from currently available data.

The impacts on avifauna are not all negative. Numbers of some species appear to increase. For example, birds that usually utilise more open areas, such as the White-winged Triller and the Rainbow Bee-eater, tend to be most common in dieback-affected forest. The Willy Wagtail and Yellow-rumped Thornbill, which are usually recorded only in open farmland and rarely in forest, were recorded in dieback Site 1.

The results of this work suggest that forest, severely affected by dieback, supports fewer species of avifauna and lower bird densities and diversity than healthy forest. Species that appear to decline include some honeyeaters and insectivores, while some other bird species characteristic of open areas tend to proliferate.

Table 3. Numbers of bird species and Shannon-Wiener Diversity (relative abundance) recorded in healthy (F) and dieback-affected (D), unmined forest sites near Alcoa's Jarrahdale (J) and Huntly (H) mines (Alcoa: unpubl. results)

	Year	FJ1	FJ2	^{A}DJ1	^{B}DJ2	FH1	FH2	^{C}DH1	^{C}DH2
Summer									
No. spp.	1992	19	13	-	8	-	-	-	-
	1995	19	16	27	7	18	18	11	14
	1998	18	11	25	12	7	11	16	14
Diversity	1995	1.99	2.42	2.19	0.69	2.42	2.41	1.74	1.88
	1998	2.26	1.96	2.44	1.63	1.59	1.73	2.15	2.24
Winter									
No. spp.	1992	18	8	-	7	18	12	21	15
	1995	20	9	22	14	17	11	13	13
Diversity	1995	1.85	0.50	1.71	1.94	2.22	1.91	1.97	1.72

Impact of disease in affected sites was ^{A}low, ^{B}high or ^{C}moderate

A second Alcoa monitoring program commenced in 1992. This again concentrated on faunal recolonisation of rehabilitated areas and it included both healthy and dieback-affected, unmined forest control sites. Monitoring was conducted every three years in both summer and winter. At each site, birds were surveyed three times along two 250m transects. All birds within 20m of the transects were counted. Results are shown in Tables 3 and 4.

Table 4. Densities (number/ha) of stated bird species recorded in 1995 in healthy (F) and dieback-affected (D) sites in unmined forest near Jarrahdale (J) or Huntly (H) (Alcoa: unpubl. data)

Species	FJ1	FJ2	[A]DJ1	[B]DJ2	FH1	FH2	[C]DH1	[C]DH2
Summer								
White-naped Honeyeater	0.17	0.33		*	*	0.33	0.33	0.33
New Holland Honeyeater				*	*		*	*
Western Spinebill		0.33	0.17	*	0.17	*		*
Winter								
White-naped Honeyeater	0.83	*	0.67	0.17	*	0.17	0.5	*
New Holland Honeyeater	*	*		*	*	*	*	*
Western Spinebill	*	*	0.33		0.33	*	*	*

*indicates that the species was present but not recorded in density estimates.
Impact of disease in affected sites was [A] low, [B] high or [C] moderate. Sites are the same as those listed in Table 3.

Considerable variation was noted between the avifaunas of healthy forest sites and this tended to mask any effects due to dieback (Table 3). Nevertheless, some trends were apparent. The low impact site at Jarrahdale (DJ1) appeared to support greater numbers of bird species than the corresponding healthy forest control sites. The reasons for this are unclear but may be related to partial opening of the canopy leading to a situation where species that inhabit open areas, for example, magpies, colonise before those requiring healthy forest are excluded.

In summer, species richness and diversity of birds at the high impact site were lower than the corresponding values for either healthy forest or the low impact site. However, the difference was not apparent in winter. No consistent differences were evident in the avifauna of forest that was moderately affected by dieback.

No honeyeaters were recorded for density counts at the high impact site in summer (Table 4). Neither the moderate nor the low impact sites showed any consistent differences in numbers of honeyeater species when compared to healthy forest sites in either summer or winter.

Other unpublished data from the 1995 study show variation in the degrees to which insectivorous species utilise dieback-affected sites. The Rufous Treecreeper was not recorded in the high impact dieback site but it was noted in a moderately affected site. The Western Yellow Robin was not found in any dieback-affected site in summer, but was present in all categories of dieback sites in winter. The Striated Pardalote showed no consistent differences between sites in any season, while numbers of the Grey Fantail tended to be least in moderately or severely affected sites in summer.

In summary, although the results have not been analysed statistically, trends indicate that in summer, severely diseased forest supported fewer species of avifauna with less diversity than healthy forest. Species thought to be affected include some honeyeaters and insectivores. The numbers of several bird species characteristic of open areas tended to increase. No declines were apparent in the low impact dieback site. In moderately affected sites, reductions in species numbers and diversity were apparent in some surveys but not in others. There were no consistent differences between the avifaunas of healthy and dieback-affected sites in winter. The large variation observed in avifaunal communities between sites with similar dieback impact indicates that more work is required to provide an adequate understanding of the interactions between bird species, vegetation changes, season, and other relevant factors.

In a separate study conducted by Wykes (1983), densities of birds in healthy and dieback-affected jarrah forest were compared. Wykes used a strip transect method similar to the two Alcoa surveys mentioned above. The dieback site was described as severely affected. Surveys were conducted every two months commencing in June 1981. Mean densities for each species are shown in Table 5 and total bird densities per count are shown in Figure 1.

Figure 1. Seasonal changes in density of bird species in healthy and dieback-affected jarrah forest (adapted from Wykes, 1983)

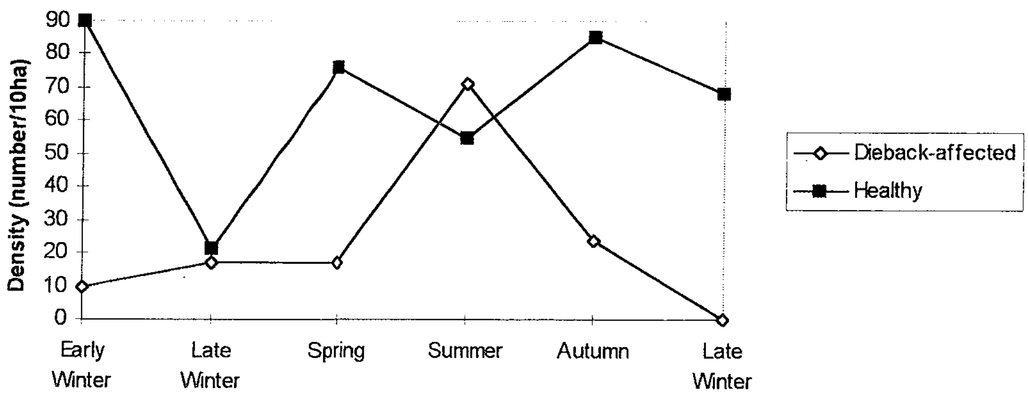

Table 5. Bird densities (number/10ha) calculated by Wykes (1983) using strip transect methods in healthy and dieback-affected sites in the jarrah forest

Guild	Species	Healthy	Dieback
Insectivore:			
a) Ground foraging	Scarlet Robin	1.9	0.9
	Western Yellow Robin	1.6	
	Splendid Wren	4.7	2.9
	White-browed Scrub-wren	0.3	
b) Shrub foraging	Western Thornbill	9.7	1.4
	Red-winged Fairy Wren	1.9	
	Grey-breasted White-eye	3.8	2.8
c) Tree foraging	Broad-tailed Thornbill	6.5	3.1
	Golden Whistler	3.3	
	Western Shrike-thrush	0.4	
	Striated Pardalote	5.8	0.8
	Spotted Pardalote	1.0	
	White-naped Honeyeater	3.6	0.6
d) Bark foraging	Varied Sittella	1.0	
e) Air foraging	Tree Martin	1.0	0.3
	Grey Fantail	6.8	1.2
	Western Flyeater	2.5	1.8
Carnivore	Little Eagle	0.5	0.3
	Laughing Kookaburra	0.6	
Nectarivore	Western Spinebill	2.1	
Graminivore	Western Rosella	0.4	0.9
	Red-capped Parrot	3.8	
	Port Lincoln Parrot	1.6	0.8
Others	seven species	0.7	0.3
Total (± s.d.)		65.5 ± 23.9	18.1 ± 18.8

Each value is the mean of six surveys conducted between June 1981 and August 1982. Total bird density was significantly greater in the healthy forest site, as was the density of the Western Thornbill, Golden Whistler and White-naped Honeyeater.

Wykes (1983) found statistically significant differences between the total bird densities at the two sites and also in the numbers of Western Thornbills, Golden Whistlers and White-naped Honeyeaters (Table 5). In all cases, densities were greatest in healthy forest. The results also suggest that the densities of many other species may be higher in healthy, than in diseased forest. Figure 1 shows that in most seasons the total number of birds were greater in healthy forest than in the site affected by dieback. The greatest differences were noted in winter.

These results accord with Alcoa data which indicate that total bird densities are generally least in areas severely affected by dieback. In both studies, densities of honeyeaters and some insectivores tended to be lower in diseased than in healthy forest and the differences were statistically significant. However, seasonal trends in bird densities do not necessarily follow those in species richness. In the work conducted by Alcoa, the difference in species richness between healthy and dieback sites was greatest in summer. In the Wykes study, the difference in density was greatest in winter.

3.3 REPTILES

The only documented information available on utilisation of dieback-affected areas by reptiles is that collected during two long term monitoring projects conducted by Alcoa. These were carried out in the same areas as the avian studies described above.

Appendix 2 shows the results obtained from sites monitored in 1981, 1987 and 1993. Methods are described in detail by Nichols & Reynolds (in prep). At healthy forest Site 1 and dieback Site 2, 10 PVC pipes (100mm deep x 150mm diam.) and 40 plastic containers (110mm depth and diam.) were installed. In addition, 10 medium and 4 large Elliott traps were used. At healthy forest Site 2 and dieback Site 1, half this number of traps were installed. To correct for the difference in sample areas, densities for each species were calculated and expressed as the number of individuals per hectare. Trapping was conducted in summer (December-January). Hand collecting and visual searches were also undertaken to obtain records of presence or absence of species. Equal periods of time were spent at each site. Although the results for sites cannot be compared statistically and must be interpreted with caution, given the variation in trap numbers, they do give some indication of whether large differences are likely to exist between healthy and severely affected dieback sites.

The total numbers of species, both recorded and trapped, differed considerably between sites and over time, but in any one year were generally highest in healthy forest. The same applied to the number of individual reptiles trapped.

Species such as the skink, *Ctenotus labillardieri*, were regularly recorded in healthy forest but never in dieback-affected sites. *C. labillardieri* is a ground forager which shelters in log or stump crevices. In any given year, several other species including *Bassiana triliniata, Hemiergis initialis* and *Morethia obscura* were usually more common in healthy, than in diseased forest. All forage amongst litter with *B. trilineata* common in shaded areas and *H. initialis* in moist sites and under logs or rocks. Other species including the skink (*Tiliqua rugosa*) and the goanna (*Varanus gouldii*) were more often recorded in healthy forest than in dieback-affected vegetation. However,

Table 6. Presence (*) and abundance (number collected) of reptile species in 1992 and 1995 in healthy (F) or dieback-affected (D) sites at Jarrahdale (J) and Huntly (H) (Alcoa: unpubl. data)

Species	FJ1	FJ2	[A] DJ1	[B] DJ2	FH1	FH2	[C] DH1	[C] DH2
Geckos								
Diplodactylus polyophthalmus			(1)	(*)				(2)
Skinks								
Bassiana trilineatum	1	(1)2	2				(1)	1
Cryptoblepharus plagiocephalus	(*)		(*)1	(1)*	*			
Ctenotus delli				(1)			*	
C. labillardieri	(*)*		(*)		(2)*	(*)3		(2)
E. napoleonis		3			1		*	(1)*
Hemiergis initialis	(1)	*			1	(3)1	1	
Lerista distinguenda	1		1	2				
Menetia greyi	1		(1)		1		2	(1)*
Morethia obscura	(1)3	(1)2	1	(1)1	(*)1	(1)3	1	2
Tiliqua. Rugosa				(*)				
Blind Snakes								
Ramphotyphlops australis	(1)						*	
Elapid Snakes								
Pseudonaja affinis Dugite								*
No. of species 1992	5	2	4	5	2	3	1	4
No. of species 1995	5	4	4	3	6	3	6	5

Results for 1992 are given in parentheses. Presence includes winter observations while abundance refers only to summer trapping.

Disease impact in affected sites (identified in Table 3) was [A] low, [B] high or [C] moderate.

the densities of some species such as the skinks *Menetia greyi* and *Cryptoblepharus plagiocephalus*, were similar in both healthy and dieback-affected forest. *M. greyi* often forages in open areas while *C. plagiocephalus*, which is commonly known as the Fence Skink, lives on logs, posts and stumps. It is often seen foraging in sunlight.

The other long term monitoring data (Table 6) were obtained using different trapping methods. At each site, traps included 10 large pit traps (150 mm diam.) with seven metre flywire drift fences, two large Elliott traps and eight medium Elliott traps. The trapping work was supplemented by hand collecting and visual searches.

Total numbers of species again differed both between sites and sampling dates, but no consistent differences were found between healthy forest and any of the dieback sites. *C. labillardieri* and *H. initialis* were more often recorded in healthy forest than in areas affected by dieback. However, at Jarrahdale, *C. plagiocephalus* was most frequently noted in dieback sites. No clear differences were apparent for other species.

Although these data are by no means definitive they do suggest a number of trends that could be investigated in more detail. These include:

- Whether total numbers of reptile species are reduced in severely affected dieback sites;

- Whether species which require litter, moist sites and shady situations are relatively common in healthy forest; and

- Whether species which forage in open areas, or on logs and stumps in exposed sunlight, are more common in areas affected by dieback than in healthy parts of the forest.

Future studies should investigate links between the presence of species and microhabitat, food, predation and competition. Predicted effects of severe dieback impacts on reptile fauna are discussed in Section 4.

3.4 FROGS

No studies appear to have been conducted on the effects of dieback-induced vegetation changes on frogs.

3.5 AQUATIC FAUNA

Horwitz *et al.* (1997) indicate that research has not been carried out on the effects of dieback impacts on aquatic fauna and suggest that changes in a plant community due to dieback may affect hydrological regimes. They also state that logging may cause hydrological changes and exacerbate the spread of dieback, thus further influencing hydrological regimes. Although recognising that the effects of such disturbances on aquatic macroinvertebrates and fish have not been investigated, Horwitz *et al.* (1997) conclude that any change in the hydrological regime would influence the composition of aquatic communities. In the absence of any detailed research it is not possible to speculate what links, if any, may exist between dieback and aquatic fauna.

3.6 TERRESTRIAL INVERTEBRATES

Several studies have investigated the degree to which terrestrial invertebrate communities differ in dieback-infected and healthy plant communities.

Postle *et al.* (1986) studied soil and litter invertebrates in healthy and dieback-infected jarrah forest. Over a 15 month period, soil and litter were sampled from 10 randomly selected points in both healthy and diseased forest. The study involved detailed measurements of litter biomass and rates of litterfall. Annual litterfall was 48% less in dieback-affected than in healthy forest. Litter biomass, total leaves, twigs/bark and fruit/flowers were all considerably reduced in the diseased area.

The abundance of most taxa was found to be relatively low in dieback-affected forest for at least part of the year and particular species were sometimes absent. The greatest differences in densities at some times of the year were noted for soil and litter Symphyla and Pauropoda, litter Araneae, Diplopoda, Coleoptera, Pseudoscorpionida and Heteroptera, and soil Chilopoda. Postle *et al.* (1986) were uncertain whether these differences were caused by changes in microclimate or by decreased availability of food. However, it was noted that proportionally more soil taxa were depleted in January and March than in any other month.

Certain invertebrate taxa were most abundant in dieback-affected sites during some seasons. These included cockroaches, ants, fly and beetle larvae and Psocopterans. It was postulated that these insects might be responding to an altered food base, but no results were provided to support this.

The conclusion to be drawn from this study is that most invertebrate taxa living in soil and litter are less abundant in dieback-affected areas than in healthy communities This is probably related to decreased litter biomass although the exact nature of the link has yet to be been determined.

Nichols & Burrows (1985) conducted a survey of predatory invertebrates in the jarrah forest as part of a study investigating recolonisation following bauxite mine rehabilitation. Predatory invertebrates were selected because it was assumed that they might reflect differences in those components of the invertebrate community which constitute their prey. The study comprised two unmined forest sites; one infected with dieback, the other free of the disease. Survey methods included pitfall trapping as well as tree beats, herb and shrub sweeps and hand collecting.

Similar numbers of spider, scorpion and ant species were recorded in both dieback and healthy sites. Numbers of centipede and earwig species were greatest at the latter site while numbers of pitfall-trapped and total species recorded were marginally lower. However, Shannon-Weiner diversity was highest in the dieback site. Abundance of herb- and shrub-inhabiting species, and all species excluding ants were greater in the dieback-affected site while the opposite was true for the abundance of all individuals.

Hamilton-Brown (1994) examined the two sites again in 1994, 13 years after the first survey was conducted. She found no large differences between the numbers of ant species in diseased and healthy areas. The number of spider species and total spider numbers were least in dieback-affected forest as were counts of total species and total number of individuals.

These results, like those of Postle *et al.* (1986), indicate that the effects of dieback-induced vegetation changes vary between invertebrate groups and between surveys. The abundance of some groups increases, possibly in response to more open ground (lower litter cover) providing better opportunities for foraging. Others such as centipedes, appear to decline. Overall, species numbers and abundance seem to decrease. For certain groups, the findings of Postle *et al.* (1986) appear to differ from those of Nichols & Burrows (1985). In some months, Postle *et al.* (1986) found that spider abundance was greatest in dieback-free forest, while Nichols & Burrows (1985) reported similar results for both sites. A possible factor contributing to the difference in findings is that dissimilar methods were used in the two studies.

The Alcoa long term fauna monitoring program discussed in Sections 3.2 and 3.3 included ant pitfall trapping in healthy and dieback-affected jarrah forest. At each site, 20 pitfall 42mm diam. traps containing Galt's solution were opened for one week in late summer and early autumn. The results from 1992 and 1995 are shown in Table 7. There was no evidence that either numbers of ant species or diversity were reduced in any of the dieback-affected sites. Distribution of individual species has not been analysed in detail. However, in 1995 it was found that *Cardiocondyla nuda*, an early coloniser that was usually apparent in the first stages of bauxite mine rehabilitation, was present in dieback affected Site DJ1, but not in any of the healthy forest sites.

Majer (1977) also conducted a study in the jarrah forest and concluded that the ant fauna of dieback-infected forest included species that were characteristic of healthy areas as well as those more commonly found in disturbed or open sites. This supports some of the findings of the Alcoa study discussed above.

Newell (1997) studied the abundance of ground-dwelling invertebrates in a Victorian forest affected by dieback. The site was located in the Brisbane Ranges, north-west of Melbourne. The study was conducted as part of the research on *Antechinus stuartii* described in Section 3.1. For the invertebrate assessment, Newell selected two low, open forest sites. The first site included areas of uninfected, recently infected and long term infected forest, while the second supported uninfected and long term infected vegetation. Fifty pitfall traps were placed in each area and sampled for one week in each of four seasons for three years. Samples were identified (usually to order) and grouped as macroinvertebrates or microinvertebrates according to the food preference of *A. stuartii*.

Table 7. Diversity and number of ant species recorded for the 1992 and 1995 Alcoa long term monitoring program in healthy (F) or dieback-affected (D) forest sites at Jarrahdale (J) and Huntly (H)

	FJ1	FJ2	^{A}DJ1	^{B}DJ2	FH1	FH2	^{C}DH1	^{C}DH2
1992								
No. species	14	14	20	19	13	19	19	22
1995								
No. species	17	7	18	18	12	11	13	14
Diversity	2.3	1.7	1.7	1.5	2.2	2.1	2.3	2.0

Disease impact in affected sites was ^{A}low, ^{B}high or ^{C} moderate.

Few statistically significant differences between the various categories of dieback-affected forest were detected for any invertebrate group. Parametrical statistical tests showed that some differences between infection groups were observed for Collembola, with higher numbers present in dieback-affected sites. In spring, abundance was highest in recently infected forest. Occasional significant differences were also observed for Coleoptera and Dermaptera, but again numbers were highest in dieback-affected sites. Few significant differences were noted for any other taxa. Non-parametric tests showed differences between infection categories for Araneae, Orthoptera, Hemiptera, Diptera and unidentified larvae on some sample dates. For all of these groups, apart from larvae, abundance was greater in infected sites than in uninfected ones. No significant differences were detected in the abundance of morpho-species of ants. Higher abundances of microinvertebrates were detected in infected than in uninfected sites for most years in both study areas.

The sampling design allowed Newell (1997) to compare interactions between seasons and dieback infection. The only statistically significant link between the two variables was found for Collembola.

Invertebrate diversity (measured using the Shannon-Weiner Index) did not differ significantly between infection categories on one site, while the long term diseased sites had the higher index of this parameter.

Detailed assessment of vegetation followed by statistical analyses were conducted to determine which factors influenced the abundance of particular invertebrate groups. Some significant relationships, both negative and positive, were found to exist at one site between the abundance of smaller invertebrates and lower structural variables such as *Xanthorrhoea australis* and litter cover. Positive correlations were found between litter cover and both Diptera and Collembola, vertical diversity and Coleoptera, and larvae and *X. australis*. Negative correlations were found between *X. australis* and Coleoptera, Collembola and Diptera, and between larvae and vertical diversity. Stepwise multiple regression analysis revealed several significant links between low vegetation parameters and invertebrate abundance, but the correlations were both positive and negative. This lead Newell (1997) to conclude that abundances within taxa were weakly associated with ground level habitat features, and therefore the impact of *P. cinnamomi* on vegetative structure and floristics was not reflected in different abundances of ground-dwelling invertebrates.

This does not necessarily mean that links between some invertebrate groups and vegetation changes do not occur. For example, a possible link existed between a temporary increase in litter cover following infection, and the abundance of Collembola. This increase in litter may be due to mortality of *X. australis*. However, the lack of any difference in litter cover between long term infected and uninfected sites may partly explain why differences in the invertebrate communities were generally insignificant.

The observation that abundance of some groups, for example, Blattodea, was correlated positively with the extent of bare ground may be an artifact of the survey method in that the mobility of individuals could be greatest on open ground, thereby increasing their chances of capture.

Summary:

Although the link between terrestrial invertebrates and dieback impact has been investigated in some detail, there are notable differences between results of the various studies. This probably reflects seasonal effects, identities of groups examined, survey methods used, identification and analyses (e.g., whether species numbers or abundance were compared) and possible differences between ecosystems (e.g., Western Australian v Victorian forests). Consistent findings are therefore few. Despite this, individual studies provide useful indications of the links that may exist.

In studies conducted in the jarrah forest, many groups appeared to decline in dieback-affected areas. These included centipedes, beetles and pseudoscorpions. This tends to result in a decrease in total invertebrate abundance. However, increases in some groups reduce the extent of this decline. Numbers of ants and ant species generally do not appear to decline, although some changes in species composition may occur with taxa more characteristic of open areas replacing those that require a healthy forest habitat.

In Victoria's Brisbane Range forest, widespread declines were not apparent. However, results of the work reviewed here suggested links between increased litter cover in recently affected sites and the abundance of Collembola. This group was also found to be more abundant in dieback-affected than in healthy jarrah forest in wetter months.

Almost all of the studies conducted to date have concentrated on ground-dwelling invertebrates. Virtually nothing is known about the extent to which changes induced by dieback might affect groups such as Lepidoptera, Orthoptera, wasps and native bees, most other pollinators and any species which feed in the canopy.

4 SYNTHESIS

4.1 IMPACTS OF VEGETATION CHANGES ON FAUNA

The studies described above are very limited both in number and in the areas and groups of fauna which they consider. None were carried out in Western Australia's southern heathlands or northern sandplains. Apart from several Victorian studies on mammals and ground-dwelling invertebrates, no work in these areas was reported for other states. In some cases, different methods were used and this limited the extent to which results for various sites could be compared.

As well as limitations imposed by the paucity of relevant studies, the nature of the disease and variation in factors relating to its spread and impact tend to complicate the measurement and estimation of dieback effects on fauna. Infested areas vary widely in the degree to which they provide habitats for particular species. This is well illustrated by considering the factors that are likely to influence impacts of dieback-induced vegetation changes on fauna. For the jarrah forest, where the majority of dieback research has been conducted, these include:

- The extent of infestation, i.e. patch size. The area infested may occupy a small part of the home range of a larger predator such as the Chuditch, *Dasyurus geoffroii*, but may cover the whole home range of a smaller mammal or bird species. Patch number and size in relation to a species' home range are likely to influence the extent of impact.

- The shape of an infested area. Narrow, linear areas such as those along stream zones are less likely to impact on some species of fauna than more rounded areas. However, the reverse may be true if the linear area coincides with the distribution of an important food species, such as *Banksia littoralis*.

- The location of infested areas. Some species may not be able to inhabit dieback-affected areas due to a lack of suitable food or shelter, or both. For these species, the extent and distribution of uninfected areas may be important because affected areas may act as barriers.

- Time since infection. The extent of vegetation and related changes vary with time. A typical sequence of impact in jarrah forest and some likely effects on fauna are illustrated in Table 8.

- Degree of impact. Some sites can be described as high impact. In these, all susceptible understorey and tree species have died, significantly changing both the structure and floristic composition of the site. In extreme cases, open ground may be present following the decomposition or burning of the litter layer. In low impact sites, the most susceptible species die but the long term survival of resistant understorey species and jarrah trees confers the appearance of low impact. More details are given in Shearer & Tippett (1989). It was previously assumed that the extent of impact could be related to Havel vegetation site type (Havel, 1975). For example, P type was considered to be high hazard, i.e. infection inevitably resulted in severe impact, while T type was considered to be low hazard. Recent observations (F. Podger, pers. comm.) have shown that this relationship is not as strong as hitherto believed Thus, the concept of hazard is no longer used in dieback management.

- Plant species present. Some animal species feed on particular plant species for part or all of the year. The Western Spinebill on *Adenanthos barbigerus* is one example. In cases where an animal is strongly dependant on one or more dieback-susceptible plant species, introduction of the disease can have a significant, local impact on the animal.

- Spread by indigenous species of fauna. It is believed that *P. cinnamomi* can be spread by species such as the Western Grey Kangaroo, but quantitative data are not available. The extent to which other native species may spread the disease is unknown.

- Spread by feral species (e.g., pigs). It has been demonstrated that pigs can spread *P. cinnamomi*. The extent to which they do this has not been quantified. The effectiveness of pig control programs is likely to influence the rate of spread.

- Predation by feral species such as the fox and cat. Feral predators have a significant impact on fauna. Whether this impact is increased in more open dieback-affected areas is not known but would seem possible. Also, it is reasonable to postulate that for some species the combined effects of feral predators and widespread dieback infestation are greater than the impacts of either in isolation.

- Floristic diversity. Following the introduction of dieback, the fauna habitat value of sites possessing predominantly Proteaceous species will be lower than that of sites which contain a greater floristic diversity including dieback-tolerant species.

- Hydrological changes. At present the link between dieback, hydrology and stream fauna is largely unknown. It is possible that if changes in stream flow, seasonality, turbidity and conductivity occur they may have some effect on aquatic fauna.

- Sub-surface drainage. The link between this and the extent of dieback impact is described in Shearer & Tippett (1989). While the general principles are well understood, in the absence of site-specific studies, it is not generally possible to accurately predict the magnitude of impact on fauna in the event of dieback introduction in a particular situation.

- Cumulative impacts. Few studies have considered the extent to which impacts of forest land uses and disturbances may be cumulative. Disturbances such as planned and unplanned fires, as well as land uses including bauxite mining and logging are all known or likely to have a temporary impact on some animal species. The total impact on a particular species will be greater if it is impacted by a number of these land uses or disturbances. Nichols & Bamford (1985) and Nichols & Watkins (1984) concluded that for birds and reptiles, the impacts of bauxite mining and dieback are generally not cumulative because the two habitats support different species.

- Regrowth of tolerant species. Marri (*E. calophylla*) eventually establishes in some areas that have been severely affected by dieback. This assists partial restoration of tree canopy cover although the process may take place over many decades. Dieback-resistant sedges also appear to be relatively abundant on many severely affected sites and they can provide ground cover for small species of fauna. The extent to which these plants spread into areas vacated by the death of susceptible species has not been measured.

- The evolutionary history of the fauna. Most of the fauna of south-western Australia are adapted to recurring disturbances such as fire and drought. This might make some species more tolerant to the types of changes that result from introduction of dieback.

In determining and managing the degree of dieback impacts, we need to first consider the nature of any given impact. These may be impacts on particular indicator species, rare species, whole communities, specific fauna groups, or a particular parameter such as species numbers, density or diversity. In the following summary of impacts, all of the above are considered.

4.1.1 Impacts on Mammals

In the jarrah forest, it is difficult to clearly define what the long term impacts of dieback changes on mammal species will be. One reason for this is that unpublished studies by Alcoa and CALM have recently shown that numbers of species such as the Chuditch and Southern Brown Bandicoot have increased significantly due to fox baiting as part of Operation Foxglove. Other species that may be increasing, or are expected to increase, include the Western Brush Wallaby, Brush-tailed Phascogale and Quokka. Reintroduction of the Woylie at a number of sites has been particularly successful, with second generation animals now being captured (P. deTores, pers. comm.). These changes in abundance make it extremely difficult to assess the extent of localised, patchy changes due to dieback.

Table 8. **Predicted sequence of vegetation changes in a typical S/P type jarrah forest (Havel, 1975) following the introduction of dieback, with likely impacts on fauna over time (It should be noted that many of the impacts are speculative.)**

Vegetation Change	Impact on Mammals	Impact on Birds	Impact on Reptiles
1. Deaths of *Banksia* and other highly susceptible species.	Decreased food resource for W. Pygmy-possum and Honey Possum.	Decreased food resource for honeyeaters and some insectivores.	Decreased food resource for arboreal geckos, temporary increase in exfoliating bark shelter.
2. Deaths of other susceptible understorey species.	Decreased shelter availability for small mammals.	Decline in wrens and White-browed Scrub-wren due to open understorey. Decline in some insectivores.	Decreased cover and shading, unfavourable microclimate. Decline in some skink species, possible increase in others.
3. Deaths of jarrah and *Allocasuarina fraseriana.*	Decreased food availability for arboreal insectivores, e.g., Brush-tailed Phascogale.	Decline in some insectivores, e.g., Rufous Treecreeper, Western Yellow Robin.	Changes in microclimate disfavouring species requiring shaded sites.
4. Increase in litter.	Possible temporary increase in food availability for ground dwelling insectivores.	Not known.	Increased food resource for some species.
5. Opening of the canopy following tree deaths.	Increased raptor predation. Change in microclimate.	Decline in some insectivores, increase in species favouring open habitat, e.g., White-winged Triller. Increased suitability for raptors.	Increased raptor predation. Unfavourable microclimate for some species.

Table 8 (Cont.).

Vegetation Change	Impact on Mammals	Impact on Birds	Impact on Reptiles
6. Possible temporary increase in hollows and logs following tree deaths.	Provision of shelter for species which require hollows and logs.	Provision of hollow nesting sites, e.g., for owls, pardalotes.	Increased shelter and foraging sites for some species, e.g., *C. plagiocephalus*.
7. Decrease in litter due to decomposition and burning.	Decreased food resource (e.g., reptiles and invertebrates).	Decreased food resource for ground foraging insectivores.	Decreased cover and food resource for ground foraging species.
8. Decrease in logs due to fire.	Decreased shelter availability e.g. for Chuditch and Bandicoots.	Possible decreased food availability for raptors.	Decreased shelter for several species ie. *Egernia napoleonis, Ctenotus labillardieri*.
9. Decrease in hollows due to trees falling.	Decreased shelter availability, e.g., for Brush-tailed Phascogale, Possum.	Decrease in hollow nest sites.	
10. Increase in marri density.	Partial increase in food availability for arboreal insectivores.	Increase in food resource for some insectivores. Seasonal increase in food availability for nectarivores.	Increase in shading and less extreme microclimate would favour some species.
11. Possible small increase in sedges, ground cover and litter.	Possible partial increase in food resource for small insectivores. More favourable microclimate.	Possible partial increase in food resource for ground foraging insectivores.	Possible increase in food resource and protection from raptors for ground dwelling species.

Certain aspects of dieback-induced vegetation changes would be expected to impact on some mammalian species. For example, tree deaths would not initially decrease the number of available hollows for the arboreal Brush-tailed Phascogale, but eventually some impact would be expected. The extent and duration of the impact would partly depend on the rate at which dieback-tolerant marri replaced jarrah. The food resource for the Brush-tailed Phascogale would also be expected to be lower in severely affected dieback areas. However, as the Phascogale also forages for invertebrates on a number of dieback-tolerant species (e.g., marri and wandoo), only localised declines would be expected.

The abundance of logs on the ground would possibly increase in some dieback-affected sites following tree deaths. It might then decline after fire. Reduced log abundance may diminish a habitat's suitability for species such as the Chuditch, Southern Brown Bandicoot and Numbat. However, decreased fox predation may lessen the species' requirement for logs to shelter from predators so the impact of reduced log abundance might not be as great.

Dieback impacts on susceptible plants would be expected to have some localised effect on some species of mammals, by decreasing nectar and (possibly) insect availability. Species such as the Honey Possum, which inhabit heathland in the Mt. Saddleback State Forest, would probably be affected if widespread deaths of nectar-producing Proteaceous plants occurred. Insectivores such as the Mardo (*Antechinus flavipes*) may be locally affected in the event of reduced abundance of suitable invertebrates. However, insufficient research has been conducted to determine the likely magnitude of any declines in food availability. Species such as the Western Pygmy-possum (*Cercartetus concinnus*) feed on both insects and nectar. In trapping programs conducted by Alcoa, both Mardos and Western Pygmy-possums have been trapped in dieback-affected forest However it is possible that these animals were itinerant individuals passing through the area. Whatever the case, the results suggest that dieback-affected sites do not form barriers to movement between remnant areas of healthy forest.

Research has shown that as a general rule, the jarrah forest's mammalian species recolonise after bauxite mining rehabilitation (Nichols, in prep.) and burning (Christensen & Abbott, 1989). Research at Kingston Block has not demonstrated either large or long term declines in the abundance of any mammal species. It can therefore be assumed that provided these and other forest activities are managed responsibly, their cumulative impacts together with dieback should be manageable.

In summary, it would seem that no mammal species in the jarrah forest is seriously threatened by dieback-induced vegetation changes, particularly in the light of increased numbers due to fox baiting. Nevertheless, some species might decline and as the extent of any such decline is unclear, the wisest course of action would be to limit the spread of dieback and manage other disturbances and land uses to minimise the overall impact of *P. cinnamomi*.

In areas other than the jarrah forest, impacts on mammalian species may be more serious. Friend (1992) speculated that the introduction of dieback into areas of southern heathland, including parts of the Fitzgerald River National Park (FRNP), is likely to cause a significant decline in the abundance of Honey Possums. This may well be correct and impacts on the Dibbler are also possible. Research investigating the likely extent of impacts would produce useful information as other values such as flora conservation and tourism are also threatened. The most urgent priority is to limit the spread of the disease in such areas.

In Victoria, sufficient impacts on small mammal species have been identified to conclude that introduction of dieback causes localised declines in faunal abundance. It is too early to say whether any species would be threatened as a result of these declines.

4.1.2 Impacts on Birds

Localised changes in the abundance of particular bird species in the jarrah forest are likely in areas impacted by dieback. These species would include honeyeaters and some insectivores. The changes would result in decline of total numbers of species in areas severely affected by dieback. Some impact on hollow nesting species might occur if the total number of hollows declines.

The Rufous Treecreeper is relatively uncommon in the jarrah forest. It forages for insects on trunks of jarrah and other trees or in log piles. The studies referred to in Section 3.2 did not record this species in severely affected dieback areas. Nor has it been recorded in rehabilitated bauxite minesites (Nichols, in prep.). Although the Rufous Treecreeper is not rare, further studies are warranted on its ecology and sensitivity to the impacts of disturbances including dieback.

The mobility of birds reduces the extent to which dieback-affected areas might act as barriers to movement between uninfected sites.

In conclusion, it is not apparent at this stage that the status of any jarrah forest inhabiting bird species is seriously threatened. However, a number of species have declined in severely affected areas, resulting in changes to the avifaunal community.

Given the absence of any relevant studies, it is difficult to make definitive statements on the likely impacts of dieback-induced changes on birds in areas outside the jarrah forest. Some rare species such as the Noisy Scrub Bird occur in areas known to be infected with dieback (e.g., Two Peoples Bay Nature Reserve; J. Blythe, pers. comm.). Others, such as the Rufous Bristlebird and Western Whipbird occur in heathland rich in Proteaceae, for example in the FRNP. The major structural and floristic changes that would follow dieback infestation in such areas could have a potentially significant impact on these rare species.

The potential impact of dieback on another rare bird species, the Western Ground Parrot, has been considered in the Species Interim Recovery Plan (Burbidge *et al.*, 1997). It was postulated that a decline in woody Proteaceous perennials could have an adverse impact on Western Ground Parrot habitat in some areas. However, the authors also recognised that such changes might actually improve the habitat by removing large

Banksia spp. and shrubs and by increasing the dominance of sedges with a possible increase in food availability for the parrot. The conclusion is that at present, the likely effects of vegetation changes on the species are unknown and further studies are required.

4.1.3 Impacts on Reptiles

The results of limited research conducted in the jarrah forest indicate that a number of reptile species are likely to decline in areas severely affected by dieback. These seem to be species that rely on an intact litter layer, moist conditions, shelter such as crevices and exfoliating bark, shade, or combinations of the above. The result of these declines is likely to be a decrease in total species numbers and reptile density in severely damaged forest areas. None of the reptile species likely to be affected is regarded as seriously threatened or at risk of extinction.

No research has been conducted on the link between dieback impacts and the reptile fauna of the southern heathlands. *Banksia* woodland north of Perth supports a species-rich reptile community. This is also likely to be true of the southern sandplains. Dieback-susceptible Proteaceae constitute a significant component of plant communities in these areas, both in terms of structure and floristic diversity. Changes to the vegetation following the introduction of dieback would be expected to significantly alter the litter, microclimate and invertebrate community with probable impacts on some reptile species. Insufficient information is available to speculate on the likely extent of these impacts on particular species.

4.1.4 Impacts on Frogs

The lack of studies carried out to date makes it impossible to provide a meaningful assessment of the impact that dieback-induced vegetation changes might have on frogs. In some situations such as the broad sandy valleys of the eastern jarrah forest, vegetation changes due to dieback may alter site hydrology and result in prolonged surface water retention. This may benefit some species which require shallow ponds, but disadvantage burrowing species of the genus *Heliοporus* due to early flooding of burrows. More work is needed to determine the nature of any impacts.

The potential of dieback-induced hydrological changes to affect rare frog species, such as the Sunset Frog, is unknown but should be investigated.

4.1.5 Impacts on Aquatic Fauna

It is not possible to speculate on the impacts of dieback on aquatic fauna beyond the conclusions already drawn in Section 3.5.

4.1.6 Impacts on Terrestrial Invertebrates

Some impacts on terrestrial invertebrates occur following the introduction of *Phytophthora*. These are most notable in severely affected areas. Reductions in litter cover and depth cause declines in numbers of centipedes, beetles and pseudoscorpions. Other groups probably decline, but the type and magnitude of changes would be expected to vary between sites and over time. A reduction in small, litter dwelling invertebrates is likely to cause a decrease in larger predatory species which feed on them.

The extent to which pollinating insects are dependent on particular dieback-susceptible plant species is unknown.

As the status of most invertebrates is unknown, it is impossible to conclude whether any species may be threatened as a result of dieback-induced vegetation changes. However, the possibility exists that some may be, particularly in areas of heathland where widespread infestation has occurred or where entire remnant patches have been infected.

4.2 MANAGING DIEBACK IMPACTS ON FAUNA

4.2.1 Constraints

Constraints that limit the extent to which the impact on fauna associated with dieback-induced vegetation changes can be managed, fall into three broad categories:

- Limitations in the amount of information available on impacts on particular species and groups of fauna.
- Limitations in the extent to which the spread of dieback can be controlled.
- The irreversible nature of changes.

Limitations in knowledge:

Clearly, there is a need for more information on the effects of dieback-induced changes on fauna. Studies to date have focused on few groups in limited areas. The extent to which dieback may threaten the conservation status of many species, particularly invertebrates, is either poorly known or unknown. Changes in the composition of bird and reptile communities are understood to a limited extent in some jarrah forest sites, but have not been studied in other ecosystems. Additional limitations in knowledge are noted in Section 3.

Limitations in controlling spread:

The application of phosphonate has proved to be an effective means of controlling the impact of disease. However, costs and the need for repeat applications may limit its use to critical areas. Spread can occur in three ways:

- Autonomous spread, which includes spread along roots, growth of mycelia, movement of zoospores and spread via ground or surface water movement. Once the disease is established control of autonomous spread is virtually impossible.

- Spread via animal vectors including native and feral species. Apart from control of pigs, which is only partially effective, limiting spread due to animals is not practicable.

- Spread due to the activities of man, specifically those that involve movement of infected soil. This is the area, where with sufficient commitment, appropriate management practices and adequate resources, further spread of the disease can be controlled.

Thus, it is apparent that only limited control of spread is possible. Introduction to new areas and autonomous spread from existing infections will continue to result in vegetation changes that will impact on fauna as discussed in Sections 3 and 4.1.

Irreversible changes:

Available evidence strongly suggests that the changes in plant communities induced by *P. cinnamomi* are irreversible. The fungus does not die out in an area after dieback has swept through and susceptible plant species have been killed. Although some re-invasion by tolerant species such as marri and various grasses and sedges may occur (Shearer & Tippett, 1989; Kennedy & Weste, 1986), plant communities on affected sites do not regain their original structure or the floristic diversity that existed prior to infection.

Some instances of susceptible species re-invading infested areas have been reported, for example, *X. australis* in Victoria's Brisbane Ranges (Dawson *et al.*, 1985). However, it is possible that the observed plants were growing in patches which had escaped infection.

4.2.2 Recommended Remedial Actions and Opportunities

There are two implications stemming from the limited availability of information on the impacts of dieback-induced vegetation changes on fauna. Firstly, research needs to be focused on animal species and groups most likely to be affected. Both management and broader research efforts need to be directed to minimisation of further spread and reduction of impacts on fauna.

Priority should be given to understanding and managing impacts on the following fauna values:

- Rare, threatened or restricted species where there is thought to be a reasonable chance of decline due to dieback-induced vegetation changes. Examples include the Dibbler, Western Whipbird and Rufous Bristlebird in Western Australia and *Pseudomys shortridgei* in Victoria.

- Species for which there is some evidence of localised decline in impacted areas and which might serve as indicators of impacts. Examples include some honeyeater, insectivore, reptile and litter-inhabiting species in Western Australia and *A. stuartii* in Victoria.

- Species and groups of fauna for which no information is currently available, but knowledge of their biology and disease impacts indicate a high probability of localised and possibly serious decline. Examples include the Honey Possum and some invertebrate pollinators such as native bees.

- Structure of faunal communities, particularly in areas where conservation is a priority land use (e.g. nature reserves and national parks).

Broader dieback research which is likely to be of significant benefit to fauna includes but is not limited to:

- Continuing research into the cost-effective application of phosphonate.

- Research into the development of dieback-resistant jarrah and other key susceptible species, including research into understanding the mechanism of resistance.

- In situations where significant populations of a rare or uncommon animal species are located and found to be at risk, there may be merit in investigating operational techniques for establishing dieback-tolerant tree and understorey species.

- Any operational research which is directed towards reducing the introduction of the disease during operations involved with land uses such as logging, mining, flower picking, beekeeping, seed collecting and tourism.

Management efforts need to be directed towards minimisation of further spread and reduction of impacts on fauna. Priority should be given to:

- Developing and implementing cost-effective procedures which will minimise the spread of the disease due to the activities of man.

- Prioritising areas that require the greatest amount of protection with respect to conservation of fauna values.

- Developing a program for planting dieback-resistant jarrah in priority conservation areas when sufficient affordable stock becomes available and where there is likely to be significant benefit to fauna and flora.

- Controlling spread of inoculum by feral species, particularly pigs.

- Controlling feral predators, particularly the fox and, where necessary, the cat (when cost effective control measures become available). This has been shown to significantly increase numbers of many species and would reduce the impacts of dieback-induced vegetation changes.

- Translocation and re-establishment of populations of some rare species known to be at threat from dieback impacts.

- Integrated studies are needed to investigate the combined impacts on fauna over time, of forest management practices such as burning, land uses such as logging and mining, and dieback. Variations on the intensive monitoring and research program being conducted at Kingston Block in the jarrah forest would be the most appropriate way to address this need.

5 OUTCOME

This document reviews available information relevant to the understanding and management of impacts on Western Australian fauna associated with structural and floristic changes to native plant communities caused by *Phytophthora cinnamomi*. Constraints on the management of these impacts are considered and remedial actions and opportunities for impact abatement are discussed. Recommendations are made for future research directions.

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Appendix 1. Mean densities of avifauna (birds ha^{-1}) in healthy and dieback-affected jarrah forest

Figures in round ( ) and square [ ] parentheses are density values from 1981 and 1987, respectively. Values not enclosed in parentheses are for 1993. Each value is the mean density derived from three surveys. Data are from Armstrong & Nichols (in prep.).

* Species observed on site in 1993 but not during survey periods.

Species	Healthy Forest		Dieback-affected Forest	
	Site 1	Site 2	Site 1	Site 2
Emu *Dromaius novaehollandiae*		*		
Little Eagle *Aquila morphnoides*		[0.33]		
Collared Sparrowhawk *Accipiter cirrhocephalus*	0.17	0.33	0.17	
Brown Falcon *Falco berigora*	*			
Common Bronzewing *Phaps chalcoptera*	[0.33]	[0.67]	1	*
Red-tailed Black Cockatoo *Calyptorhynchus magnificus*	*		[0.67] 0.83	[0.5] 0.33
Baudin's Cockatoo *Calyptorhynchus baudinii*		1.17		
Australian Ringneck *Platycercus zonarius*	(0.67) [0.33] *	0.5	0.33	0.33
Red-capped Parrot *Platycercus spurius*	(1.5) [0.83] 0.33	(0.5) [0.5] 0.17	(0.67) 0.83	(1.17) 0.5
Western Rosella *Platycercus icterotus*	(2.5) 0.67	[0.17] *	*	(0.17)
Horsfield's Bronze Cuckoo *Chysococcyx basalis*		0.17		
Laughing Kookaburrra *Dacelo gigas*	*	(0.34) [0.67] 0.17	0.17	0.83
Sacred Kingfisher *Halcyon sancta*	[0.17]	*	(0.34)	
Rainbow Bee-eater *Merops ornatus*		*	(1.84)	(3.17)
Tree Martin *Cecropis nigricans*		(1.84) *	[0.33]	(1.33)
Black-faced Cuckoo-shrike *Coracina novaehollandiae*	*			(0.17)
White-winged Triller *Lalage sueurii*				(0.83)
Scarlet Robin *Petroica multicolor*	[0.33] *	0.17	(0.34) [0.33] *	[1] *
Western Yellow Robin *Eopsaltria griseogularis*	(0.84) 0.83	(0.34) [0.17] 0.5	*	
White-breasted Robin *Eopsaltria georgiana*	[0.83]	[0.5]	[0.5]	[0.33]
Golden Whistler *Pachycephala pectoralis*	(0.17) 0.33	[1.5] *		0.33

Appendix 1 (Cont.).

Species	Healthy Forest		Dieback-affected Forest	
	Site 1	Site 2	Site 1	Site 2
Rufous Whistler *Pachycephala rufiventris*	0.17	[0.17] 0.5	*	0.33
Grey Shrike-thrush *Colluricincla harmonica*	(0.5) [0.17] 0.5	(0.17) [0.17] 0.33		
Willie Wagtail *Rhipidura leucophrys*			*	
Grey Fantail *Rhipidura fuliginosa*	(0.84) [0.5] 0.83	(1) [1.83] 0.33	[1] 0.33	[0.5] 0.83
Weebill *Smicrornis brevirostris*	(0.17)	(0.34)	(0.34) [0.17] *	
Western Gerygone *Gerygone fusca*	(0.17) [1] 0.33	(0.17) [0.17] 0.83	(0.67) [0.33] 1.33	(0.33) [0.83] 0.17
Inland Thornbill *Acanthiza apicalis*	(0.67) [0.83] 0.17	(0.34) 0.5	[0.33] 1	(0.17) [0.33] 0.83
Western Thornbill *Acanthiza inornata*	(2.17) [1.5] 4	(0.5) [0.5] 1.83	(0.67) [0.67] 0.5	(1) [1.83] 1.67
Yellow-rumped Thornbill *Acanthiza chrysorrhoa*			*	
White-browed Scrubwren *Sericornis frontalis*				[0.17]
Red-winged Fairy-wren *Malurus elegans*	[0.17] 1		[0.17]	0.5
Splendid Fairy-wren *Malurus splendens*	[0.5]	(1.17) *	[0.33] 0.33	
Varied Sitella *Daphoenositta chrysoptera*	*	(1.5)		(0.83)
Rufous Tree-creeper *Climacteris rufa*	(0.67) 0.33	(0.17) [0.33] 0.33		
Striated Pardalote *Pardalotus striatus*	(0.84) [1] 0.33	(2.17) [1.33] 0.67	(0.5) [0.83] 0.83	(2.5) [0.5] 0.5
Spotted Pardalote *Pardalotus punctatus*	(0.5)			
Silvereye *Zosterops lateralis*	[0.17]	*		(1.17)
Brown Honeyeater *Lichmera indistincta*		[0.17]		
White-naped Honeyeater *Melithreptus lunatus*	(0.5) [0.17] 0.67	(1.17) [1.17]		
New Holland Honeyeater *Phylidonyris novaehollandiae*		[0.67]		
Western Spinebill *Acanthorhynchus superciliosus*	(0.5) 1.33	(0.84) [1.5] 0.67		(0.5) [0.17] 0.83
Little Wattlebird *Anthochaera carunculata*	0.33	*		(0.17)
Dusky Woodswallow *Artamus cyanopterus*			(0.67)	
Australian Magpie *Cracticus tibicen*	[0.67] *		[0.5] 0.5	*
Grey Currawong *Strepera versicolor*	*	*		
Australian Raven *Corvus coronoides*	*	0.33	*	

Appendix 2. Numbers of reptiles per hectare in healthy and dieback-affected jarrah forest

* Survey 1=1981, 2=1987 and 3=1993.

+ Species observed but not trapped. Results are from Nichols & Reynolds (in prep.).

Species	Healthy Forest						Dieback-affected Forest					
	Site 1			Site 2			Site 1			Site 2		
***Survey**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**
Gekkonidae												
Diplodactylus polyophthalmus	8			16								
Phyllodactylus marmoratus	+											
Agamidae												
Pogona minor (minor)	+						64					8
Varanidae												
Varanus gouldii	+	+		+	+		+			+		
Scincidae												
Bassiana trilineata	+	24	8		16	48		32			8	8
Cryptoblepharus plagiocephalus	+	16	16		32	16	32	32		40	16	16
Ctenotus delli	+		8									
Ctenotus labillardieri	16	8	8	32	32	48						
Egernia napoleonis	32		+		16	32	+	16			8	
Hemiergis initialis	+	8	24	32		16			16		8	+
Lerista distinguenda	112	48	24	480	32	48	32		208	64		104
Menetia greyii	24	24	56	256	16	16	48	16	16	32	32	8
Morethia obscura	352	176	48	224	32	80	64	64	112	16		32
Tiliqua rugosa	32		+	+	16		16					+
Elapidae												
Notechis scutatus	+											
Pseudonaja affinis	+		+					+				
Rhinoplocephalus nigriceps				+								
No. of species recorded	16	8	11	9	9	8	8	6	5	5	5	8
No. of species trapped	7	7	8	6	8	8	6	5	4	4	5	6
No. of individuals trapped	576	304	192	1040	192	304	256	160	352	152	72	176

CONTROL OF *PHYTOPHTHORA* AND *DIPLODINA* CANKER IN WESTERN AUSTRALIA

FINAL REPORT

TO THE THREATENED SPECIES AND COMMUNITIES UNIT

BIODIVERSITY GROUP

ENVIRONMENT AUSTRALIA

DECEMBER 1998

Department of Conservation and Land Management
Locked Bag 104, Bentley Delivery Centre, WA 6983