THE IMPACT OF DISTURBANCE ON TERRESTRIAL INVERTEBRATES IN THE WESTERN AUSTRALIAN RFA AREA

A Report to the Commonwealth and Western Australian Governments for the Western Australian Regional Forest Agreement

by

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Authors Note

Due to time constraints, this project has necessarily been completed without the usual opportunity for a lengthy process of critical reflection on, and digestion of, the data. Consequently, the review, while accurately mirroring the author's viewpoints, may be deficient in regards to the presentation normally associated with a published work. However this may be, the authors stand by their interpretations and conclusions.

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Note

The authors have indicated that in their view additional time for this project may have enhanced the quality of this report. The Commonwealth and Western Australian Governments welcome any further information or comment from these authors on the matters dealt with in this report, or on additional matters within their area of expertise relevant to the Regional Forest Agreement, within the public consultation period.

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INTRODUCTION

The Concept of Ecosystem Health

Here, in Australia, we are only beginning to emerge from a world view of our natural ecosystems as entities that are designed for our exploitation, or as alien forces that must be controlled and subdued. It is still not uncommon, unfortunately, to hear political leaders complaining that largely pristine areas have not been developed, as though lack of commercial exploitation of 'resources' in these areas is something reprehensible. However, we are coming to a gradual realisation that things are profoundly wrong with our natural surrounds. In south-western Australia there is increasing evidence of human abuse of the environment: e.g., salination resulting from clearing of woodlands and shrublands; the spread of dieback fungus (*Phytophthora cinnamomi*); proliferation of feral plants and animals; and the pollution of waterways by toxic metals (among other pollutants). The question to be addressed is: 'How can we act responsibly and proactively to changes we are imposing on our natural environments?'

There have been various attempts to evaluate factors that can be used as indicators of unhealthy ecosystems. Ideally, suitably trained environmental managers will be able to recognise these indicators and recommend suitable remedial action. Schaeffer, Herricks and Kerster (1988) use the analogy of the diagnosis of human disease to show how poorly we understand the health of our ecosystems. They suggest that the following are symptoms of profound ecosystem ill-health:

- Falling numbers of native species;
- Large-scale ecosystem change in the direction of earlier stages of succession;
- Changing crop biomass;
- Changing gross or net primary energy production;
- Changing relative amounts of energy flow to grazer and decomposer food chains;
- Changes in mineral macronutrient stocks; and
- Changes in both the mechanisms of, and capacity for, damaging undesirable oscillations.

Criteria for healthy ecosystems must be established cautiously, and with due regard for the ecosystem being evaluated. Variability in an ecosystem may reflect no more than response to short-term and ephemeral environmental conditions. Alternatively it could be due to malignant and irreversible changes. The above authors suggest the following criteria that could be considered as a basis for initial assessment of ecosystem health:

- A diagnosis of ecosystem health should not be based on a single species;
- Health should not depend on a census or inventory of large numbers of species;
- The diagnosis should reflect our knowledge of normal succession or expected sequential changes which occur naturally in ecosystems;
- Ecosystem health does not have to be measured as a single number;
- Ecosystem health should assume that the measurements used to assess it have a defined range;
- Ecosystem health measures should have a single value (i.e. be monotonic) and vary systematically in a discernible manner;
- The health criteria should be responsive to change in the data, but should not show abrupt change;
- Health measures should have known statistical properties;
- Criteria for health assessment must be related, and hierarchically appropriate for use in ecosystems;
- Health measures should be dimensionless or share a common dimension, so that components can be added or compared on a common basis; and
- Health measures should be insensitive to the number of observations, given some minimum number of observations.

The importance of Invertebrates as Indicators of Environmental Health

Maintenance of fundamental ecological functions is crucial to the creation of self-sustaining ecosystems, whether it be a harvested forest, a plantation or an area of rehabilitation. Table 1 shows the various ecosystem functions which need to be maintained, or in the case of drastically disturbed areas, re-established, and shows some of the outcomes and typical organisms which are involved. Animals, particularly

Table 1. The various ecosystem functions which should be established in rehabilitation, the outcomes and typical organisms associated with each ecosystem service.

ECOSYSTEM FUNCTIONS	PEDOGENESIS	DECOMPOSITION	MYCORRHIZAL ACTIVITY	PHOTOSYNTHESIS	HERBIVORY	POLLINATION	PROPAGULE DISPERSAL	PREDATION/ PARASITISM	OTHER SPECIES INTERACTIONS (mutualists, symbionts, competition)
	⇓	⇓	⇓	⇓	⇓	⇓	⇓	⇓	⇓
OUTCOME	improves soil structure	cycles nutrients	enhances nutrient uptake, some stimulation by soil fauna	provides cover and diversity, contributes water cycling	limits excessive togrowth, enhances plant diversity	enables reproduction of plants	enables dispersal of plants	limits pests, enhances animal diversity	Limits pests, enhances plant and animal diversity
	⋮	⋮	⋮	⋮	⋮	⋮	⋮	⋮	⋮
SOME ORGANISMS WHICH ARE INVOLVED	ants, termites, worms	millipedes, springtails, mites, worms	soil microarthropods	lichens, ferns, mosses, higher plants	beetles, caterpillars, kangaroos	wasps, flies, birds	ants, birds, mammals	wasps, flies, beetles, reptiles	ants - nectaries, mite - domatia

invertebrates, play an important role, so much so that they have been referred to as 'the little things that run the world' (Wilson 1987). The importance of invertebrates is such that they are regarded by some as the 'drivers' of ecosystems, whereas most of the vertebrates can be considered to be 'passengers' (Walker 1992).

Most of the species of animals alive today are invertebrates; depending on whose estimates are used, the number of extant insect species alone ranges from 8-100 million (Groombridge 1992). In terms of biomass, invertebrates usually exceed vertebrate animals in the same area (New 1995). Ants alone are believed to constitute about 30% of global terrestrial fauna biomass (Holldöbler and Wilson 1990, p. 1). Yet there is usually no problem in justifying the outcome of research dollars on vertebrate animals. The low profile of invertebrates in forest management and restoration research is totally unjustified in view of their importance in ecosystem functioning.

This discussion paper reviews the impact of disturbance on terrestrial invertebrates within the Western Australian Regional Forest Agreement (RFA) envelope. It concentrates on studies performed within this envelope although, where relevant, studies from nearby areas such as the Swan Coastal Plain or the Wandoo woodlands are included. In one case where no relevant studies on a particular disturbance and taxonomic group were available, we have drawn upon information from outside the State.

SCOPE AND APPROACH OF THE REVIEW

The Taxonomic Impediment to Progress

The taxa which we discuss under the various types of disturbance are inevitably dictated by the scope of work which has been carried out in this State. Some groups, such as ants, termites and spiders have been reasonably well investigated; taxonomic and life-history studies have been conducted on the local fauna, and specialist expertise is available to the ecologist. Consequently, ecological papers, especially in more recent years, have been able to provide detailed and finely resolved information on the impact of disturbance on such taxa. Information on the impact of disturbance on others, such as orthopterans, is scanty and, on the majority of groups, it is non-existent. This is hardly surprising in a country with conservatively 2-300,000 invertebrates (Coy 1995). The true number is likely to be much higher: as an illustration of this, recent work on the cicadellid subfamily Typhlocybinae has raised the known Australian species from 15 to 168, i.e. an astonishing 1100% (Mann and Fletcher 1997). Insects, however, fare rather well compared with our knowledge of groups like mites and nematodes.

Approach adopted for this Report

Despite the difficulties posed by our poor knowledge of many invertebrate groups, there is an increasing literature on forest invertebrates in Western Australia (see bibliographies by Majer and Chia, 1980 and by Abbott *et al.* 1986). We only refer to these papers where they relate to disturbance.

Most of the above papers have looked at the impact of disturbance on insects and the larger arachnids (mainly spiders). This is predictable, as insects and spiders, compared with some other groups, are relatively easier to find in numbers than are many other invertebrates. Moreover, they can often be collected by traditional methods, and have a tolerable familiarity (usually at the Order, and sometimes at the Family level, as with ants) for student researchers and other non-specialists. This does not necessarily mean that insects and spiders are always the best bio-indicators. Such groups as earthworms include potentially vulnerable species with very specialised requirements and localised distributions. Several papers by Abbott (1982, 1985) on earthworm species occurring in and near the RFA area are discussed here.

We first consulted the published literature and apportioned the papers into the types of disturbance listed in Table 2. The fifth and last columns of the Table ('*Phytophthora*' and 'Pest outbreaks') are actually more of an outcome of disturbance than an impact.

Table 2. Summary of eleven different kinds of disturbance in the RFA area, with literature on the various terrestrial bio-indicator groups.

Taxon	Logging	Plantations	Mining	Fire	Phytophthora	Agriculture	Urbanisation	Recreation	Damming	Introduced species	Pest outbreaks
Order or various	Norwood et al 1995	Abbott 1993 Majer et al 1997	Majer 1989a, 1989b, 1990a, 1990b, 1992, 1997 Nichols et al. 1989 Nichols & Burrows 1985 Casotti 1987	Whelan & Main 1979 Whelan et al 1980 Main 1981 Friend 1996 Friend & Williams 1996	Newell 1997 Nichols & Burrows 1985					Majer 1994	Abbott, 1992a, 1992b Abbott, Van Heurck & Williams 1993 Abbott et al. 1993 Fox & Curry 1980 Mazanec 1980, 1988
Soil and litter arthropods		Majer 1977 Fitzgerald 1994	Majer 1977, 1981 Ward, Majer & O'Connell 1991	McNamara 1955 Abbott 1984 Friend 1995 Friend & Williams 1996 Hindmarsh & Majer 1977 Little & Friend 1993 Majer 1977, 1984, 1985 Springett 1976, 1979	Majer 1977 Postle et al. 1986	Abbott et al. 1979 Abensperg-Traun, Arnold et al. 1996 Abensperg-Traun, Smith et al. 1996	Majer 1977				
Canopy arthropods	Recher et al. 1996 Springett 1978	Springett 1971, 1973, 1976				Majer, Recher & Keals 1997					
Spiders	Curry et al 1985		Simmonds et al. 1994 Mawson 1983	Strehlow 1993 Main 1981							
Collembola			Greenslade & Majer 1980, 1993 Nichols et al. 1989								
Termites			Bunn 1983 Nichols & Bunn undated Nichols et al 1989	Abensperg-Traun & Milewski 1995 Abensperg-Traun, Smith et al 1996		Abensperg-Traun 1992	Postle & Abbott 1991				
Orthoptera				Dolva 1993							
Ants		Burbidge et al 1992 Majer 1977, 1985	Majer 1977, 1981, 1983 Majer & Beeston 1996 Majer & Nichols 1997 Majer et al. 1984 VanSchagen 1986 Nichols et al 1989 Scott 1980	Majer 1977 Boardman 1985	Majer 1977	Majer & Beeston 1996 Scougall et al. 1993	Majer 1977 Majer & Beeston 1996 Majer & Brown 1986 Burbidge et al. 1992	Burbidge et al. 1992	Woodroff & Majer 1981	Davis et al. 1993 Anon. 1988	
Earthworms	Abbott 1985		Abbott 1985								

Because of their significance from the invertebrate point of view we have included them in this discussion paper. The papers were also segregated by the taxon or taxa which had been investigated. A small portion of the papers concentrated on individual species, families or orders. Several papers covered soil and litter invertebrates or canopy invertebrates so we erected a category for both of these. Finally, most of the papers either considered invertebrates in general at the ordinal level or reviewed the impact of disturbance on a range of invertebrate animals. These were placed in the first, 'unspecified' row of Table 2.

THE DISTURBANCES

1. Forestry Practices

What happens to these animals when the forest is harvested, managed or cleared? What happens when attempts are made to rehabilitate disturbed areas of forest? The type of forest operation has a profound influence on the how, which, and how many animals are affected. In the following sections, which are general rather than specific descriptions, the influences of the major forest disturbances on forest fauna are discussed. The main changes in habitat associated with each practice are identified and the ways these might impact on the fauna are described.

a. *Rotational clear-felling*

This type of harvesting removes all or most of the trees once they have attained a certain level of maturity (Figure 1). Trees are cut and removed, resulting in considerable disturbance to the soil and litter and temporary destruction of the understorey. Seed trees, which provide the next generation, may be retained, although these are often cut once they have released their seed. The cut area may be burnt at this stage to reduce trash and to stimulate seed release. Following this reduction of a layered forest of considerable structural complexity, there commences a regrowth of even-aged trees. The resulting forest tends to be homogeneous and it may take a considerable period of time for the understorey to reach its former complexity.

Responses of fauna to this type of disturbance can be due to the reduction of overstorey in the early stages of regrowth - many of the soil and litter organisms depend on the shady conditions that a mature forest provides. These organisms may also be affected by the burn, if this is carried out (see section **f.**). Other organisms are dependent on particular strata of vegetation in the forest or on large, mature habitat trees - if these are not present these animals will not occur or may be less abundant in the regrowth. The appropriate habitat eventually builds up again but, after certain rotational period, the forest is logged once again, thus resetting the succession. The degree to which this type of logging affects the conservation status of species depends on the proportional extent of the operation. If most of the forest is logged then vulnerable fauna will be adversely impacted, if an adequate proportion of the forest is left unlogged then refuges for these species will probably be adequate.

b. *Selective logging*

This type of logging is only directed at mature trees or trees of economic significance. Areas which contain such trees are identified and accessed by tracks or by directly driving into the forest (Figure 2). The required trees are cut and removed, sometimes with resulting damage to nearby trees, the understorey and the soil layer. The result of this is that tree mortality can be higher than the number of trees that are logged (Johns 1992). Some of the non-merchantable trees may be cut or poisoned at this stage in order to reduce competition with the remaining commercial trees. The outcome of this type of logging is a forest of reduced canopy density containing fewer habitat trees. The understorey may become more dense in response to the increased light regime and invasive species may enter along tracks or in response to the disturbance. Where present, pathogens such as dieback (*Phytophthora cinnamomi*) may be spread by

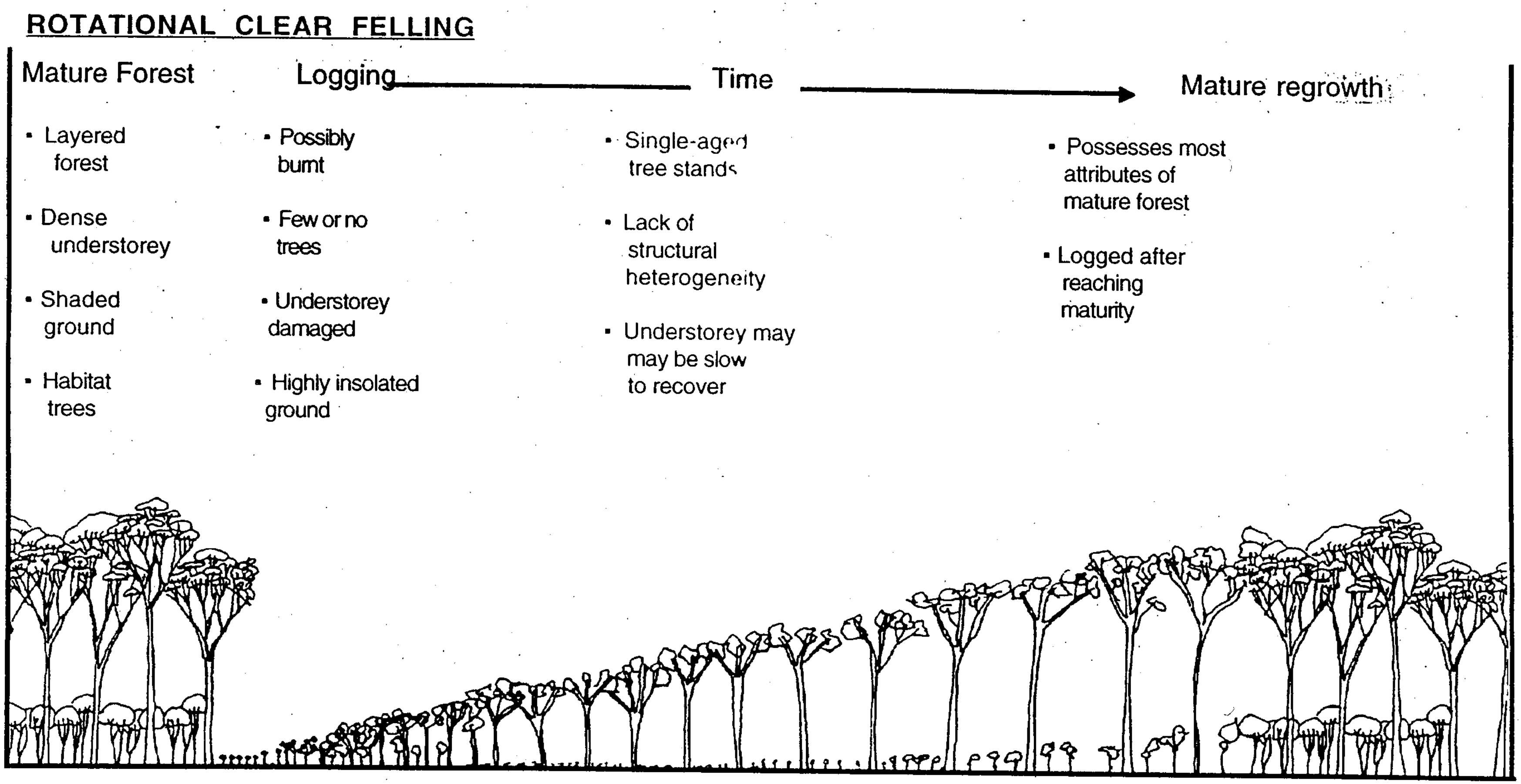


Figure 1. Rotational clear-felling showing gradual recovery of the forest over time.

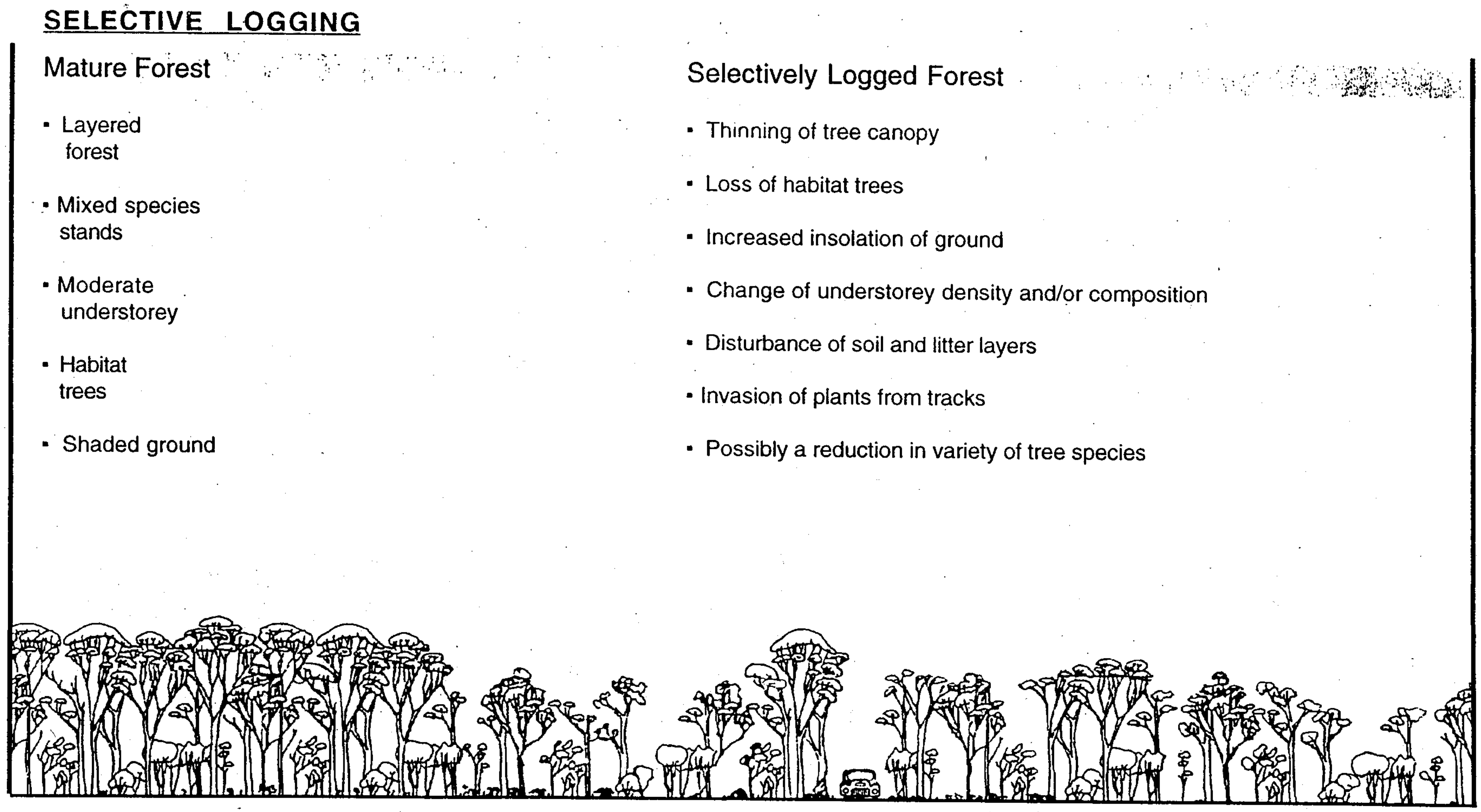


Figure 2. Selective logging. Note the alteration to composition and density of canopy and understorey (centre and right of figure).

vehicular access, further exacerbating the changes in the habitat. Following the logging event, the area is revisited and logged when more of the commercial trees have reached an adequate size.

The response of the fauna is probably not as great as for clear-felling. If the variety of tree species is reduced then there may be a loss of those herbivorous insects which are specific to certain tree species. Weed invasion may competitively reduce the native plant species in the understorey and this, too, could lead to the loss of species-specific herbivores and their associated parasites. If habitat trees are not retained, the birds and other vertebrates which nest in them may be reduced. There is evidence that the disturbance to the ground layers by vehicles and tree felling may pose a considerable threat to some of the mammals and other vertebrates that inhabit the area. If the forest becomes infected by dieback, or if it simply takes a considerable time to regrow, the changed nature of the habitat may lead to long-term changes in the characteristics of the forest fauna.

c. ***Gap creation***

A variation on selective logging is to identify areas of mature trees and to clear-fell patches within the matrix of unlogged forest (Figure 3). The area may or may not be burnt to remove trash, and a limited number of habitat trees are sometimes retained within the patches. Areas where logging is carried out are systematically rotated around the region so that the forest contains a mosaic of regenerating patches of varying age. This is sometimes referred to as gap creation and the procedure can result in track construction, increased insolation of the ground, increases in understorey density and stands of even-aged trees.

Faunal responses can be dramatic during the initial stages, with invertebrates responding to the changed understorey and birds responding to the extensive ecotones that are created around the edges of the patches. Roading can lead to spread of invasive

GAP CREATION

Mature Forest

- Layered forest
- Mixed species stands
- Moderate understorey
- Habitat trees
- Shaded ground

Logged Gap

- Temporary absence of tree canopy
- Loss of habitat trees
- High insolation of ground
- Thickening of understorey
- Area may be burnt
- Ecotones created
- Invasion of plants from tracks
- Possibly a reduction in variety of tree species

Maturing Forest

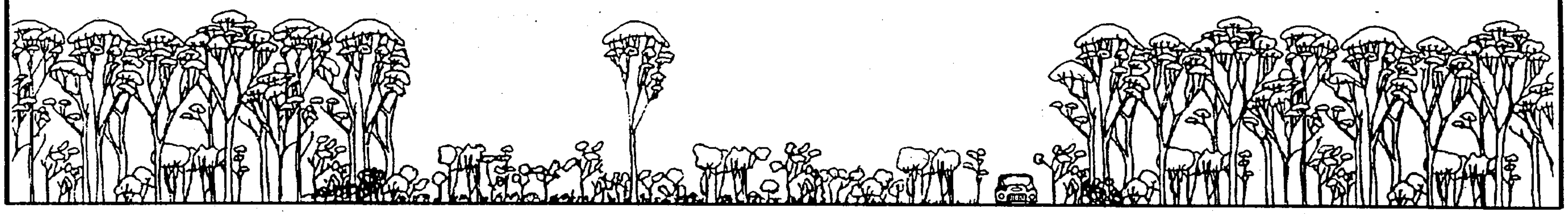


Figure 3. Gap creation showing temporary loss of tree canopy. The regrowth (right) may lack the floristic composition of the original forest.

plants, resulting in further changes in the original understorey and its associated invertebrate fauna. As with the previous logging practice, there is always a danger of the spread of forest disease and a simplification of tree diversity if non-commercial trees are cut out.

d. *Plantation establishment*

Native forests are being converted to mono-specific plantations of trees that are generally not native to the area (Figure 4). The most commonly used trees are *Pinus radiata* and *Eucalyptus globulus*, although *Pinus pinaster* has been used on the coastal plain. The plantations lack the structural complexity of the original forest and, as a result of their often allelopathic effects, tend to have extremely simplified understoreys. The impact on the native fauna is massive. By lacking the horizontal and vertical stratification of the original forest, the area fails to provide habitat for most of the original vertebrate inhabitants. The lack of food resources has an even more pronounced effect. Importation of plants into totally new areas leads to an association with herbivores which have little or no adaptation to feed upon that species. The result is that these trees tend to be barren of insects, which themselves are food resources for much of the bird fauna. One of us (JDM) has witnessed an area in Brazil where one of the last remnants of Atlantic rain forest had just been felled. The area was surrounded by *Eucalyptus* plantations and the felled forest was covered by encircling birds looking for food and habitat. Few birds were seen in the plantation.

An additional problem which threatens plantations is their vulnerability to pest outbreaks (Schowalter and Means 1988). This can result from their lack of evolved defence to particular herbivores or to the paucity of beneficial limiting agents. Large tracts of *Eucalyptus* in southern Brazil have been totally defoliated by leaf-cutter ants (*Atta* spp.), even though the vegetation of the surrounding areas is largely unaffected (JDM pers. obs.). The result is destruction of those features of the plantation that do provide habitat for local fauna.

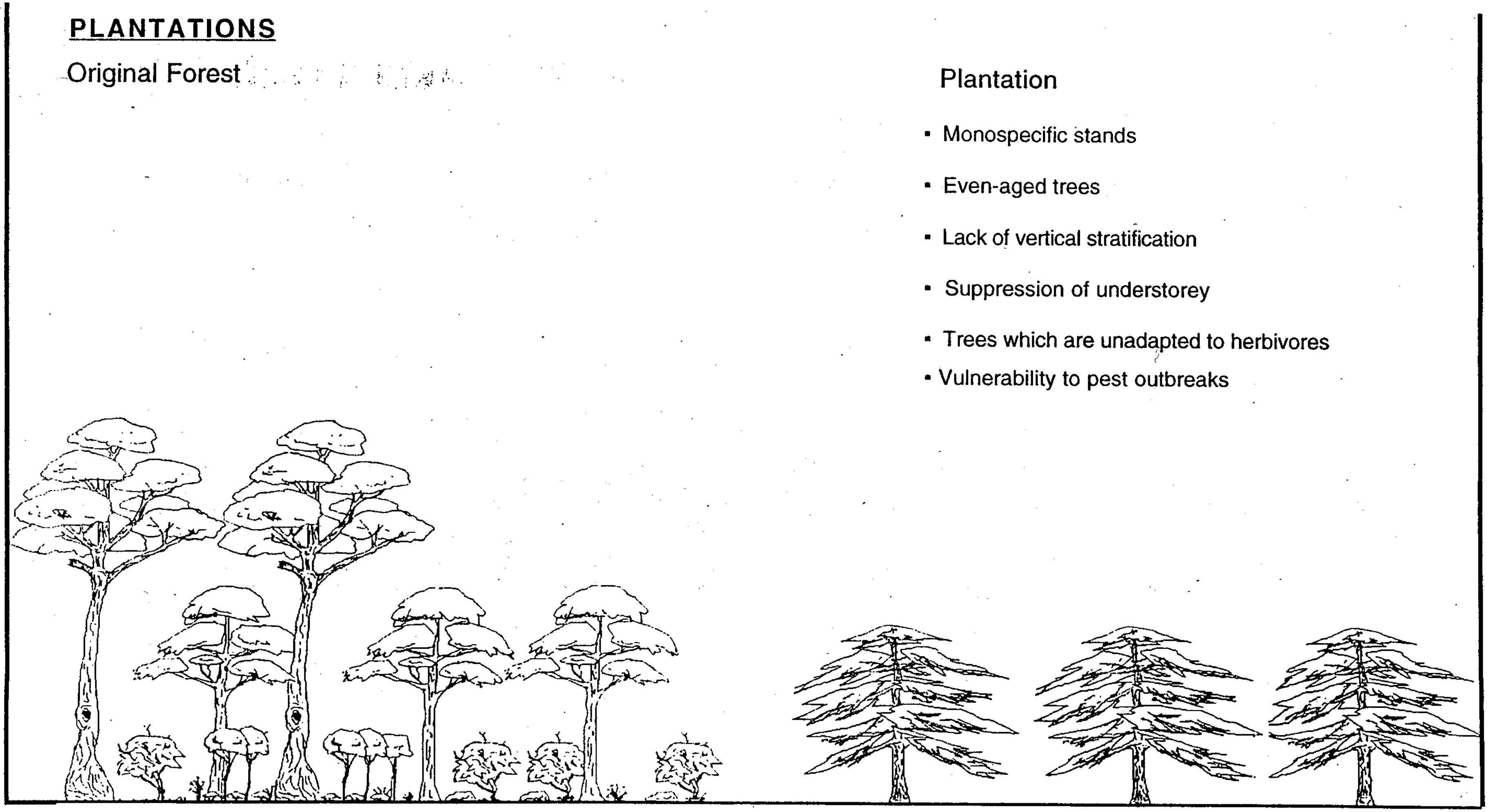


Figure 4. Creation of a plantation (right side of figure). Note the monospecific stands of even-aged trees, with an absence of understorey vegetation.

e. *Agroforestry*

Agroforestry operations attempt to combine tree production with grazing by stock on grasses or forbs in the understorey (Figure 5). The forest may consist of native or introduced trees; whichever is the case, the trees are generally grown at a reduced density in order to allow rapid growth of the understorey. The understorey is continuously or intermittently grazed by stock and is generally of a considerably lower structural and species diversity than that of the original forest. The original understorey may have been replaced by introduced grass species.

If the area consists of an introduced tree species then the comments of the last section naturally apply. However, the major impact of this type of operation is on the animals associated with the understorey. The wide range of invertebrates associated with the different plant species are undoubtedly curtailed. Those birds and other vertebrates which depend on particular plant strata will probably be absent if these strata have been removed.

The presence of stock can have additional adverse effects on the conservation status of the ecosystem. Trampling results in soil compaction, which renders the soil inhospitable to many soil organisms. The excreta of these animals elevate soil nutrient levels, resulting in high foliar nutrient levels. This can result in high herbivore loads, sometimes to outbreak levels that can result in death of the tree. Thus the presence of such animals can have further deleterious effects on the conservation status of agroforestry ecosystems.

A specific example of agroforestry is the commercial exploitation of honeybees *Apis mellifera* for honey production. These animals are not native to certain forested regions, such as Australia, and compete with native bees for nectar. In addition to the

AGROFORESTRY

Thinned Forest with cows grazing understorey

- Reduced vertical stratification
- Reduced canopy density
- Understorey simplified - probably replaced by grass
- Soil enriched by nutrients and compacted

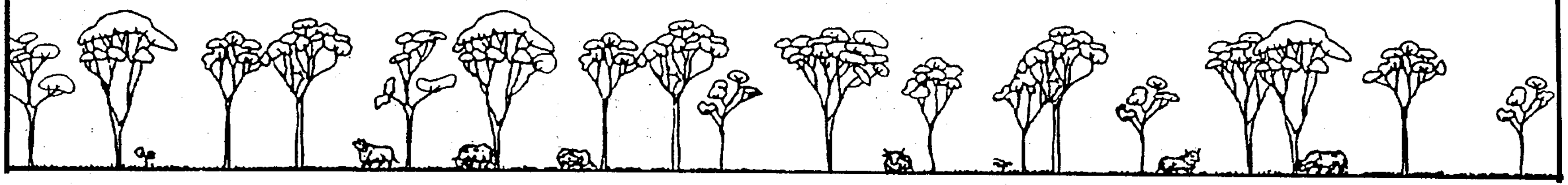


Figure 5. Original forest converted to agroforestry operation. The canopy has been reduced and the understorey is now mainly grass.

conservation threat to native insects, this can also lead to break down in species-specific pollinating systems.

f. *Burning*

Forests in some of the drier regions of the world are deliberately burnt at intervals of time in order to reduce accumulated organic material and hence reduce the fire hazard (Figure 6). The immediate effect of the fire is removal of the litter layer, scorching of the understorey and, depending on the intensity of the fire, some of the tree canopy, plus death of certain plant species. Such forests are generally considered to be fire-tolerant and subsequently recover as result of germination by obligate seeders and resprouting from epicormic shoots and subterranean plant parts. Thus the effect of such fires is diminished as the forest recovers.

The impact on animals of prescribed burning in drier forests has been the focus of considerable research. The litter invertebrate fauna is radically altered in the short term, both in terms of abundance and diversity. Similarly, the shrub-associated invertebrates are eliminated by the burn, although they recolonise as the vegetation regrows. Colonisation is much slower in large burnt areas where the distance to sources of animals is greater. Some vertebrates, such as reptiles and amphibians, are killed by the fire, while others such as birds and mammals often move away. Once again, the larger the fire, the less opportunity there is to escape. Views differ on whether, and how rapidly the invertebrate fauna recovers in the years following a fire. There is some evidence that, with many forests now being burnt so regularly, the entire forest fauna may have been altered by burning. This is resulting in an almost total absence of unburnt bench marks upon which to measure the influence of burning.

g. *Forest clearing*

Throughout the world, and particularly in the tropics at present, forests are being

BURNING

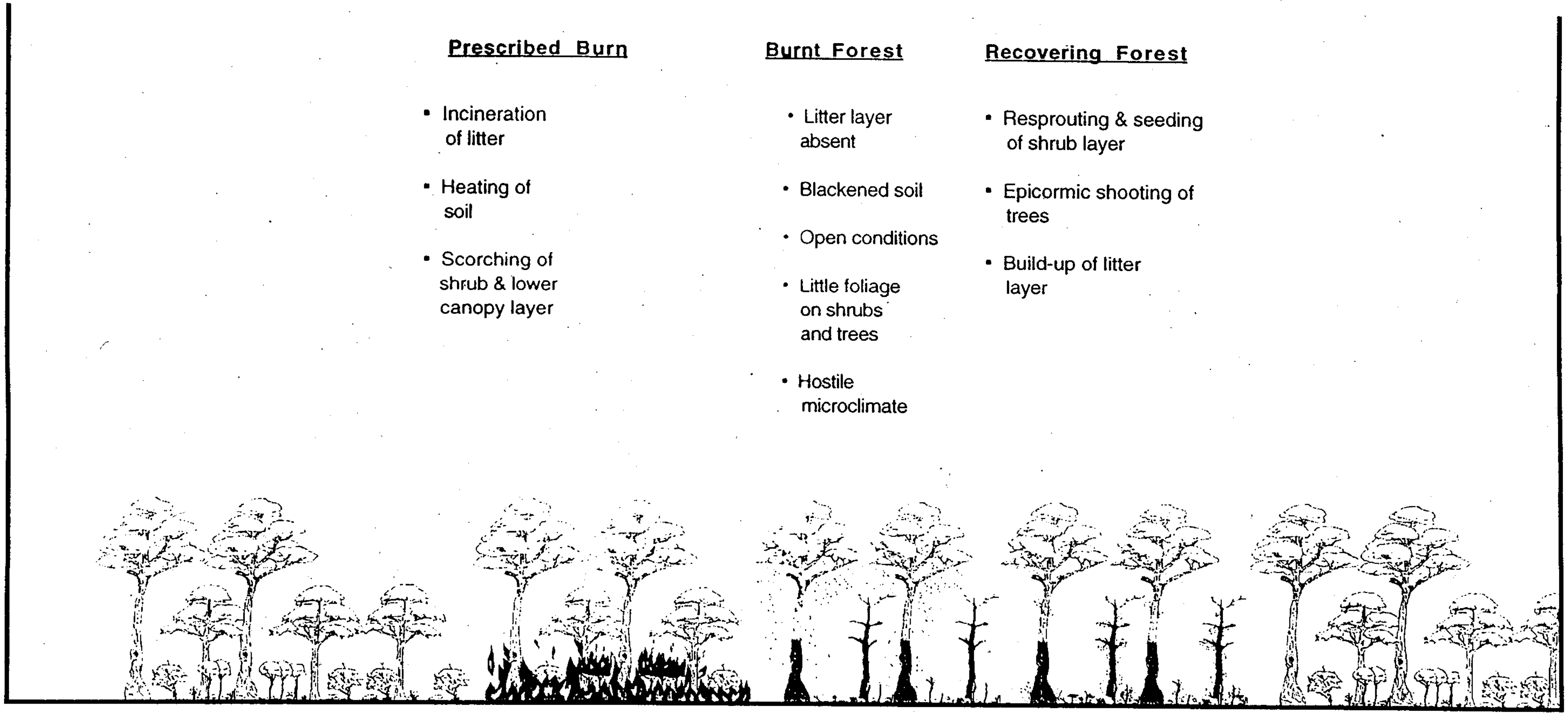


Figure 6. Fire in a fire susceptible forest (right). Although forests in the southwest of Western Australia are predominantly fire tolerant, regular burning regimes may be resulting in the simplification of the fauna.

cleared to make new land for agriculture (Grainger 1992). This may be planned, or it may occur in an *ad hoc* way following the opening up of areas by tracks, sometimes the very ones that have been installed for exploitation of the forest. One new land use which follows clearing, particularly of rain forest at the present time, is rangeland. Clearing is often accompanied by burning, so the area experiences many of the impacts of burning mentioned in the last section (**f**). The impact is further magnified by the presence of stock which compact the soil, enrich it with nutrients, and graze the understorey.

An additional outcome of both types of agriculture is that the remaining forest is becoming fragmented into smaller and smaller patches (see centre of Figure 7). Often the size of the fragment is insufficient to support viable populations of some plants and animals. Additionally, pronounced edge effects exist, with changes in microclimate and floristics at varying distances into the fragment. A further degenerative influence is wind; vegetation is no longer buffered against wind, so trees in the remnant tend to blow over or snap off, thus resulting in clearings within the fragment.

Fauna are affected by agricultural clearing. The impact of burning in fire-susceptible forests has already been discussed in section **f.** and the impact of stock in forested lands has been outlined in section **e**. The combined effects of deforestation, burning and stock in the created farmlands means that the area experiences an almost total change in the original fauna. The fragmentation of the forest by these farming activities places added stress on the fauna. Not only do animals suffer loss of habitat, but they are adversely affected by edge effects, which for different groups of organisms extend to varying distances into the forest. The general degradation of these small and vulnerable areas of forest creates additional problems. It is estimated that by the year 2000, more than half the tropical forests in the world will have been cut, and much of that which remains will be affected by fragmentation and other degrading influences. This will result in the extinction of as many as 600,000 species, most of which are animals (Laurance and Bierrogaard 1997).

CLEARING FOR AGRICULTURE

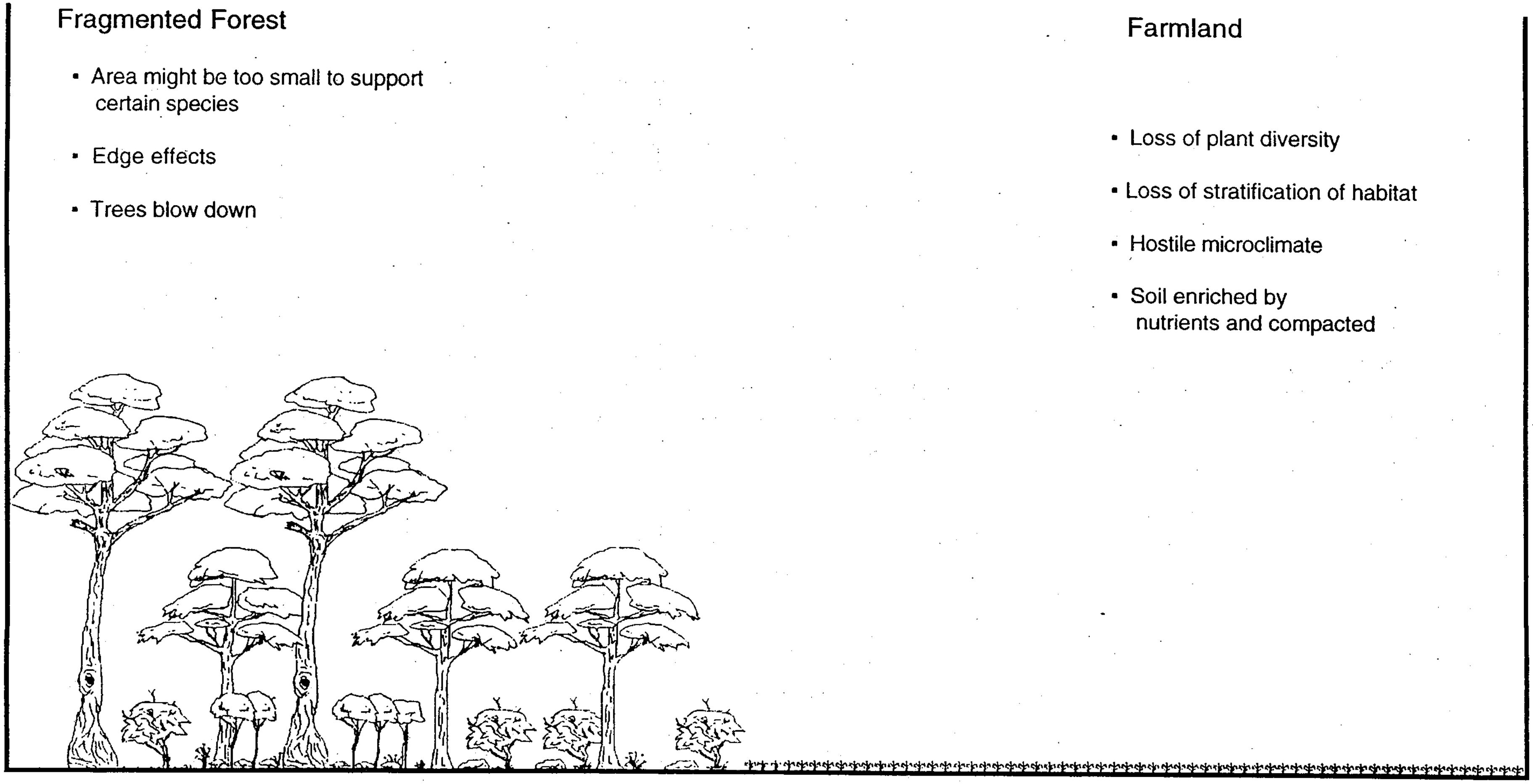


Figure 7: Jarrah forest to cropping

h. Other disturbances

The perturbations outlined above are the result of deliberate forestry practice. However, a number of disturbances that can affect forests, such as *Phytophthora cinnamomi* infestations (Figure 8) and urbanisation (Figure 9) are not the result of deliberate forestry practice. Indeed, disturbances can be quite unforeseen, and in some cases may only become evident a number of years after they were first initiated.

2. Specific Disturbances in the RFA area

Eleven results of human activity that have impinged on, and continue to affect, the RFA area are now described. The amount of attention that has been given to particular types of disturbance varies greatly: for example, there are upwards of a score of invertebrate research papers devoted to fire and mining in the jarrah and karri forests and surrounding areas, but less than half-a-dozen on invertebrate pests in the same areas. Only one paper (Woodroff and Majer 1981) discusses the effect of dams on forest invertebrate fauna, and there are no detailed studies on the impact of vehicular traffic (including trail bikes and other off-road vehicles) and horse-riding on invertebrate fauna. Eco-tourism is another activity of growing importance that has not been covered in any research paper on forest invertebrates.

In this section there is a brief overview of the available literature on a particular type of disturbance, then individual papers, reports and book chapters are discussed and their findings summarised. A subjective evaluation of the literature is contained in Table 3. (In this Table the difference between '*' and '**' is somewhat arbitrary, but in general, the one asterisk denotes a smaller research programme. Often this is a 'one-off' study done by a student, or a preliminary survey concerned mainly with quantitative data. Two or three asterisks denote generally longer programmes extending for more than a year, sometimes with experimental manipulation of variables.) From this Table, also, general theoretical reviews, simple inventories and purely natural history papers have been omitted, although they are they are discussed in the text. The quality of research is

Table 3. An evaluation of the literature covering disturbances in or near the RFA area. Key: 'o' superficial research with confused or inaccurate taxonomy; '*' preliminary survey, findings usually confined to abundance and richness of target taxa; '**' more comprehensive research programme carried out over more than one year, usually several variables examined; '***' monographic treatment of a single taxon with a comprehensive analysis of the influence of one or more environmental variables on that taxon over a period of more than one year.

Author(s)	Disturbance									
	Logging	Plantations	Mining	Fire	*Phytophthora*	Agriculture	Urbanisation	Damming	Introduced species	Pest outbreaks
Abbott (1984)				*						
Abbott (1985)	**		**							
Abbott *et al.* (1993)										**
Abbott, Parker & Sills (1979)						*				
Abbott, Van Heurck & Burbidge (1993)										**
Abensperg-Traun (1992)						*				
Abensperg-Traun & Milewski (1995)				*						
Abensperg-Traun et al. (1997)						**				
Abensperg-Traun, Arnold et al. (1996)						**				
Abensperg-Traun, Smith *et al.* (1996)						**				
Abensperg-Traun, Steven & Atkins (1996				*						
Anonymous (1988)									*	
Boardman (1985)				**						
Bunn (1983)			*							
Burbidge *et al.* (1992)		*					*			
Casotti (1987)			o							
Curry *et al.* (1985)	**									
Davis *et al.* (1993)									*	
Dolva (1993)				***						
Fitzgerald (1994)		*								
Fox & Curry (1980)										*
Friend & Williams (1996)				**						
Greenslade & Majer (1980)			**							
Greenslade & Majer (1993)			**	*						
Hindmarsh & Majer (1977)				*						

Author(s)	Disturbance (cont.)									
	Logging	Plantations	Mining	Fire	*Phytophthora*	Agriculture	Urbanisation	Damming	Introduced species	Pest outbreaks
Majer (1977)		*		*	*					
Majer (1984)				*						
Majer & Brown (1986)							*			
Majer & Nichols (1997)			**							
Majer *et al.* (1984)			**							
Majer, Recher & Keals (1997)						**				
Mazanec (1980)										***
Mazanec (1988)										***
McNamara (1955)				*						
Newell (1997)					**					
Nichols & Bunn (undated)			*							
Nichols & Burrows (1985)			*		*					
Norwood *et al.*(1995)	*									
Postle, Majer & Bell (1986)					**					
Recher, Majer & Ganesh (1996)	**									
Scougall, Majer & Hobbs (1993)						**				
Scott (1974)			o							
Simmonds (1994)			*							
Simmonds, Majer & Nichols (1994)			*							
Springett (1971)		**								
Springett (1973)		**								
Springett (1976)		**		**						
Springett (1979)				*						
Strehlow (1993)				**						
van Schagen (1986)			*							
Ward, Majer & O'Connell (1991)			**							
Whelan & Main (1979)				**						
Whelan, Langedyk & Pashby (1980)				*						
Woodroff & Majer (1981)								*		

briefly evaluated, especially for those areas where there is disagreement. Finally, a group or groups of forest invertebrates that may have value as indicators of disturbance are identified from the existing literature.

a. *Logging*

The literature on impact of logging on invertebrates in the RFA area is slight. Only one paper directly examines a group of arthropods (Curry *et al.* 1985), though Recher, Majer and Ganesh (1996) and Recher and Majer (1996) discuss arthropods of the eucalypt canopy. A paper by Abbott (Abbott 1985) on the indigenous earthworms of the northern jarrah forest has relevance to fire, logging, mining and plantations. This paper is discussed below under 'Mining'. Norwood *et al.*'s 1995 article on the effect of edge and gap creation and bird populations also describes an invertebrate sampling exercise done to establish likely food sources for insectivorous birds. However, the findings, which were tabulated as abundance figures based on sweep-netting only, are superficial. Springett's (1978) theoretical paper on the ecological role of insects in Australian eucalypt forests indirectly touches on the issue of logging. This author concludes that phytophagous
insects are very important players in a nutrient recycling scheme that sees them skim off nutrients from a slow nutrient cycle (maintained by the trees) and transfer them to a faster cycling pool (in the litter and understorey). Clearly, operations such as logging, which can affect this delicate balance, could potentially have a considerable effect on rate regulation of grazing, and consequently on overall forest health.

Curry *et al.* (1985) found evidence that clearing (and burning) wrought marked changes in an arachnid community in karri forest. Diversity was highest in mature forest and lowest after clearing. Species richness recovered from the disturbance more slowly on creek sites than on ridge sites. Unfortunately, since the karri area was burned nine to fifteen months after being clear felled, not one but two variables (i.e. logging and burning) are involved in this study. This makes it harder, even impossible, to evaluate the effect of logging alone on the arachnid community.

The two papers by Recher and Majer (1996), and Recher, Majer and Ganesh (1996) respectively, deal with the one study. Canopy arthropods collected from a jarrah/marri forest at Karragullen in Western Australia were compared with narrow-leaved ironbark and grey box canopy arthropods from Scheyville, New South Wales. While this study did not examine the effect of disturbance directly, the paper by Recher, Majer and Ganesh (1996) concluded that eucalypt forest ecosystems may show relatively low resilience following disturbance. Consequently, a reduction in arthropod species richness may therefore have serious long-term implications for forest management. These researchers nominate the loss of less abundant parasites and predators as especially important.

Clearly, the literature on the effect of logging on invertebrates in the southwest forests is minimal. This is difficult to understand, particularly in view of the long-standing nature and economic importance of the timber industry. Curry *et al*'s (1985) work is the only one which details possible effects of logging on a taxonomic group, and as has been seen, this study combines the effects of two types of disturbance. This is unfortunate, since the taxa analysed by Curry *et al.* were identified to genus level. Thus, they could have provided more useful data than many other ecological surveys on invertebrates that treat ordinal level only.

b. *Plantations*

The literature on the effect of plantations on invertebrates in the RFA area is also meagre. Three of the five studies reported on here were 'one-off' student surveys of ant populations (i.e. Majer 1977, 1985; Burbidge *et al.* 1992). Another was a third year student project comparing litter invertebrate faunas of native karri forest and a Tasmanian bluegum (*Eucalyptus globulus*) plantation (Fitzgerald 1994). The fourth study involved a much more extended research program on soil microarthropods by J. A. Springett (Springett 1971, 1973, 1976). In addition there is a brief review by Abbott (1993) of intercropping theory in plantations, and its effect on certain guilds of invertebrates. His views have yet to be tested on eucalypt plantations.

The two papers by Majer detail sampling of epigaeic invertebrate fauna (but principally ants) at Dwellingup and Dryandra respectively. The 1977 Dwellingup study involved pitfall trapping of seven sites in jarrah/marri forest. Three of the sites represented different states of jarrah forest (i.e. unburnt, burnt and dieback-infested), an old town site now reverted to scrub, a mined area, farmland and a *Pinus radiata* plantation; hence the examination of fauna in a plantation was only a minor component of this study. The ant species from the plantation were found to be most similar to those of the townsite and the farmland. Fewer ant species were collected on the plantation than in the other six sites. The 1985 study had two phases. The second phase examined the effect of fire on invertebrates, and will be discussed in section **d**. The first phase, like the preceding study, was a preliminary sampling of ant fauna in 14 plots, only one of which was a *Pinus radiata* plantation. Pitfall trapping was again used as a sampling method, but daylight hand collections were also performed. Ant species richness was lower in the pine plantation than in the other 13 plots.

The above studies reveal a depauperate ant fauna in the pine plantations at the time the studies were performed. Additionally, the 1977 study was carried out in a 15-year-old pine plantation which had not been burned, indicating that the ant fauna had not returned to a composition comparable to that of the three jarrah areas during this time. One ant species (*Iridomyrmex conifer*(Forel)) was collected in enormous numbers in the plantation. The second study, at Dryandra, does not supply any details concerning the nature of the pine plantation.

The study by Burbidge *et al.* (1992) also involved ants. The research was conducted at Yanchep National Park, north of Perth. A variety of native plant communities (e.g.

Dryandra heath, *Banksia* woodland, jarrah woodland, and tuart woodland) were sampled. Gardens, two eucalypt plantations and two pine plantations were also sampled. The plantations and gardens exhibited the most altered ant communities, reduced species and generic-level richness, and fewer ant functional groups. Using an ordination analysis, the species composition of ants on the pine plantations was shown to be quite distinct from that of ants in the eucalypt plantations and gardens, as well as in the other sites. The researchers concluded that plantations are dominated by one or a few species.

Fitzgerald (1994) sampled leaf litter invertebrates from karri forest and from a plantation of Tasmanian bluegum using Tullgren funnels. Although there were found to be significantly higher numbers of some invertebrate orders in the karri forest on a per unit area basis, the density and weight of litter per unit area was greater for karri than for Tasmanian bluegum. Therefore, if this is taken into account, the invertebrate values were not significantly higher in the karri forest. Despite this, Liddelow (1993) found that leaf litter decomposed at a slower rate under the bluegum plantation than under the karri.

Springett's earliest paper (Springett 1971) describes the effect of a controlled burn in a *Pinus pinaster* plantation at Gnangara in 1966. She found that the decomposition rate was much reduced in the burnt area, and the proportion of fungal decomposers was lower. The 1973 study gives far more biological detail for the target groups (principally Acarina, Pseudoscorpionida and Collembola) than any of the other papers mentioned. Six sample sites were chosen, five of which were under *Pinus radiata* and one principally under native vegetation. Only one of the *Pinus* sites is recorded as having been burned. Fewer micro-arthropods with food in their gut were found on the burnt stand than on the other sites. This site also had the lowest decomposition rate. Overall, the number of rare microarthropods was lower in the pine plantations than in the site under native vegetation. The 1976 study follows the same lines as the preceding study, except that four sample sites were chosen (three under *Pinus*, one under native vegetation). The findings were also the same as in the preceding study.

Abbott's (1993) theoretical review is written from a agroforestry rather than a conservation perspective. Most of the case-studies cited involve crop plants or eastern Australian eucalypts rather than native South-west forest trees, and in general this review has little applicability in the present context.

The limited nature of the plantation studies outlined above suggests that much has yet to be done to determine the effect of plantations on a range of invertebrates. There has been no published local research into the effect on invertebrate fauna of plantings of Australian natives, such as *Eucalyptus globulus*, nor has the effect of plantations on arachnids and major insect groups, excepting Formicidae, been documented. Furthermore, apart from Springett's studies, there has no been ongoing research involving experimentation with different variables within plantations (e.g. fire history, age stands or soil profiles). The data on ants, however, consistently show an altered species composition as well as reduced biodiversity for pine plantations compared with undisturbed sites or plantations of native species.

c. ***Mining***

The literature of mining is voluminous compared with that for the previous types of disturbance. A number of papers are case studies (i.e. Abbott 1985; Bunn 1983; Casotti 1987; Greenslade and Majer 1980, 1993; Majer 1977, Majer and Nichols 1997; Majer *et al.* 1984; Nichols and Bunn undated; Nichols and Burrows 1985; Simmonds 1994; Simmonds, Majer and Nichols 1994; van Schagen 1986; Ward, Majer and O'Connell 1991). There are also several general reviews (Majer 1989a, 1989b, 1992, 1997, Nichols, Majer and Wykes 1989), special area reviews (Majer 1978a, 1980, 1983, 1990a, 1990b, 1997), two reports (Majer 1978b, 1981), and student theses (Mawson 1983; Scott 1974).

The case studies document research using six taxonomic groups or communities as bio-indicators, i.e. ants (Majer 1977; Majer *et al.* 1984, Majer 1997; Majer *et al.* 1984, Scott 1974; van Schagen 1986), arachnids (especially spiders; Mawson 1983; Simmonds 1994; Simmonds, Majer and Nichols 1994), Collembola (Greenslade and Majer 1980, 1993), earthworms (Abbott 1985), predatory invertebrates (Casotti 1987; Nichols and Burrows 1985) and termites (Bunn 1983; Nichols and Bunn undated). Several of these studies also provide lists of arthropods identified to ordinal or family level, though little is done with these lists.

(i) *Ants*: Studies on the West Australian ant fauna prior to the early 1980's reflect the inadequate knowledge of this group prior to Greenslade's (1979) publication, which gave a guide to the ant genera of South Australia. Scott's (1974) thesis is particularly deficient, with inaccuracy compounding a poor taxonomic coverage of the ant fauna of a Jarrahdale bauxite area. Majer's (1977) article on the ant fauna of Dwellingup gives a more satisfactory coverage of the fauna, though only one of the seven sites sampled was a bauxite mine. This survey was preliminary only, but revealed that the species composition of the bauxite mine was midway between townsite and farm sites and jarrah areas in terms of similarity of species composition.

van Schagen's (1986) study is much more comprehensive than the previous two, and demonstrates an increased sophistication in analysis of physical and plant parameters. Four methods (pitfall trapping, hand collection, vegetation sweeping and tree beating) were used to collect the ants. More information is given concerning the plots, which comprised ten rehabilitation sites and two forest controls. van Schagen found that ant species richness was positively correlated with plant species, but negatively correlated with soil penetrability. Ant species richness was associated with time. Ant succession data are provided in this paper, though most species were still designated only by voucher numbers. (The bulk of these ants are described species whose names are now known.) As well as ants, other arthropods are listed in this paper, and identified to the ordinal or family level. No sophisticated analysis of these arthropods was attempted, but the total number of species and taxa was greater in the control plots than in the mined areas.

Majer *et al.* (1984), and Majer (1997) examine the ant fauna of a range of rehabilitated bauxite minesites in Dwellingup and Jarrahdale. Majer *et al.* (1984) surveyed 30 bauxite mined sites in summer. They found that ant return was positively associated with plant species richness and diversity, time since rehabilitation, percentage plant cover and percentage litter cover. Plant species richness and diversity, rehabilitation age and percentage litter cover also influence the species composition of the ant community in rehabilitated areas. Majer's (1997) paper supplies updated information on ant succession based on continued monitoring of rehabilitated minesites and forest controls at Dwellingup and Jarrahdale. Probably the most significant finding was that a local meat ant, *Iridomyrmex greensladei*, was common on unvegetated and planted plots of more than three years, but was uncommon in the seeded plots and forest. This ant, and other meat ant species, are known to affect species composition and diversity of other ants occurring close to their nests (Greenslade 1976, Andersen and Patel 1994).

(ii) *Arachnids*: Mawson's (1983) BSc (Hons) thesis gives an account of research on arachnids done in bauxite mines and surrounding forest at Jarrahdale. Pitfall trapping, litter sampling and unit-area visual searches resulted in the capture of 85 species of arachnid (comprising 79 species of spider, 2 Opilionida and 4 scorpion species). The arachnid fauna of three rehabilitated bauxite minesites was compared and contrasted with that of two jarrah sites. All sites had differing histories. Mawson measured species diversity, evenness and richness, and concluded that at least eight years was required for the habitat structure of a rehabilitated bauxite minesite to be capable of supporting a arachnid community comparable with that of surrounding undisturbed jarrah forest. In an appendix (Appendix 5) Mawson also made recommendations for the improved management of rehabilitated mines so that the habitat would be more amenable to colonisation by arachnids. These recommendations included addition of suitable rocks, logs and a leaf litter layer on the sites, use of low intensity fire (on a site with dense

Acacia canopy) to open up the tree canopy and reduce compaction of the leaf litter layer, variation in the types of plants and seedling trees used for rehabilitation, and the planting of native annuals to increase productivity of the sites.

The papers by Simmonds (1994) and Simmonds, Majer and Nichols (1994) report on the same research programme, also conducted on bauxite mines at Jarrahdale. Since this study was intended to complement Mawson's work, the same methods used by Mawson were, for the most part, adopted here. The research, however, was limited to Araneae. Although 151 species were taken, only 44 of these were collected in both studies. The researchers corroborated Mawson's findings that older rehabilitated sites had a richer spider fauna. Leaf litter depth and cover, as well as vegetation density, had a significant positive influence on recolonization by the various spider guilds. Moreover, spider communities were found to change as vegetation matured on the older sites, resulting in a switch from pioneer species to Araneae that required less harsh conditions.

(iii) *Collembola*: The two papers by Greenslade and Majer (1980, 1993) cover the same research programme, conducted in 1978-1979. However, the earlier paper reports only on data obtained from three rehabilitated plots and one forest control plot. The data for the full number of rehabilitated plots (30) and forest control plots (three) are analysed and discussed in the later paper. Greenslade and Majer (1993) stated that overall, 68 species of Collembola were collected, 28 from the forest plots and 60 in the rehabilitated plots. Nine of the 28 forest plot species were not present in the mine sites. Principal components analysis suggested that species richness of the collembolan community in rehabilitated areas is positively correlated with plot age. The researchers also found that the development of a species rich collembolan fauna is positively correlated with plant species richness and diversity, and with percentage plant cover.

(iv) *Earthworms*: Abbott (1985), conducted a study on the indigenous earthworms (Family Megascolecidae) of the northern jarrah forest between 1979 and 1983. Samples were collected by hand from blocks of soil collected at random along a 100 m transect, and from soil-litter cores from the same 15 by 15 m^2 plot. He found that mining for bauxite or gravel was the most adverse disturbance in respect of earthworms. Clearing and reforestation with pines had a variable impact (an adverse effect in one case, but nil effect in two other cases). Logging had no impact on earthworms, but fire had a minor direct and some indirect effects on these animals. (A slightly puzzling feature of this paper is that the different species, admittedly probably mostly undescribed, are not given any other than 'morpho-species' status. However, an earlier paper (Abbott 1982) is able to discuss a number of indigenous earthworms of the Perth area at genus or even species level.)

(v) *Predatory invertebrates*: Nichols and Burrows (1984) studied predatory arthropods (Arachnida, Chilopoda and the insect orders Hymenoptera (Formicidae only), Dermaptera, Blattodea, Mantodea and Coleoptera) in bauxite mined areas in the Darling Ranges, south of Perth. (No exact locality or localities is/are named in this work, but Casotti's 1987 article indicates that the research took place at Jarrahdale.) The purpose of the study was to assess the suitability of different rehabilitated areas for providing the habitat requirements of predatory invertebrates. Unfortunately, this study is limited in a number of respects (e.g. very little data on the four rehabilitated sites, a dieback affected area and a healthy forest plot are given, the duration of the study was only December 1980 to January 1981 without follow-up, and the taxonomy is restricted to Order for Insecta, and Family for the other groups). The authors concluded that areas rehabilitated using outdated techniques possess lower species richness and diversity than healthy or dieback affected forest. They recommended the introduction of logs, a leaf litter substitute (chipped wood waste), and elimination of roads around pits to promote a favourable environment for a predatory community similar to that found in unmined forest.

Casotti's research (Casotti 1987) was intended as a follow-up of the efforts of Nichols and Burrows. Even more than its predecessor, this publication is a slight piece of work, and furthermore, the taxonomy is defective: Solpugidae, an extralimital arachnid group, are shown as having been found in all four study sites! In addition, the insect taxa listed are an odd assortment of Orders and Families. Casotti concluded that species richness had increased in all areas (except for one rehabilitated site seeded with *Eucalyptus muellerianna*) and invertebrate diversity had also increased.

(vi) *Termites*: Bunn (1983) identified 14 species of termites from Willowdale, 110 km south of Perth. He considered the termite *Amitermes obeuntis* Silvestri to be particularly important for minesite rehabilitation, because its mounds are frequently inhabited by at least five other species of native termites. The paper by Nichols and Bunn (undated) concluded that almost all termite species found in native jarrah forest at Jarrahdale returned to rehabilitated mined areas. These researchers also recommended that forest practices which ensure that colonies of early colonising species such as *Coptotermes acinifomis raffrayi* are established as quickly as possible.

Four of the six special area reports by Majer (i.e., Majer 1978a, 1980, 1990a, and 1990b) discuss the role of ants in Australian land reclamation seeding projects. The reports are relevant to the theme of disturbance insofar as some ant species will predate on propagules used in seeding rehabilitated minesites. This may interfere with the restoration of native flora in disturbed areas. Ant genera of the jarrah forest known to be seed-takers in rehabilitated areas are *Iridomyrmex*, *Melophorus* and *Rhytidoponera* (Majer 1990a). Seed removal occurs up to 25 m into the mined area (Majer 1990a). Majer (1983) argues for the use of ants as bio-indicators, and alludes to his Dwellingup survey (Majer 1977) and to other surveys in the southwest and Pilbara districts.

The two reports (Majer 1978b, 1981) cover the same research programme; a continuous monitoring pitfall trap programme, carried out mostly at Del Park (Dwellingup). A forest control plot and planted and unplanted minesites were involved. A seeded area was located at Jarrahdale. Majer (*op. cit.*) showed that the unplanted mine plot supported the least variety and the lowest number of invertebrates. The variety of invertebrates was similar in the planted and seeded plots, but invertebrate numbers were much higher in the latter. The total ant number, ant biomass and ant species richness increased from the unplanted plot, through the planted plot to the forest control plot. However, ant species richness did not attain the forest control plot values in any mine site.

The articles by Majer (1989a, 1989b) and Nichols, Majer and Wykes (1989) are chapters in the volume entitled 'Animals in Primary Succession' (ed. Majer; Cambridge University Press). Majer's chapters are general reviews with a global focus on vertebrates as well as invertebrates, while the chapter by Nichols, Majer and Wykes looks more narrowly at the return of vertebrates and invertebrates to bauxite mined areas in southwest Australia. Majer's (1997) article is a book chapter in 'Restoration Ecology and Sustainable Development' (ed. Urbanski; Cambridge University Press). In this article Majer reviews work done by Australian researchers on rehabilitated lands. All of these articles mention various of the case studies discussed above.

Taken overall, a review of the literature on mining in the southwest suggests that a good case can be made for the use of ants, Collembola, spiders and several termite species as bio-indicators of rehabilitation success on minesites. The use of earthworms as bio-indicators is more problematic, not least because of the difficulty of accurately identifying native species about which probably nothing is known. Use of guilds (such as predatory invertebrates) as bio-indicators has problems of complexity that need to be taken into account in any interpretation of results. More will be said of this later. Thus far, almost all studies have been confined to two or three localities in the northern jarrah forest. Research also needs to focus on minesite rehabilitation in karri and other climax vegetation types in the south-west

d. *Fire*

There is a plethora of papers on fire, far more, in fact, than on any other aspect of forest disturbance. Some of the most well-researched articles on fire in the southwest of this state relate to the influence of fires in woodlands and shrublands adjacent to the RFA area. These will also be discussed here. In addition, many other articles mention fire as one of several components involved in forest disturbance.

The scope of the fire literature relevant to the RFA area ranges from simple faunal surveys carried out shortly after a fire event, to monographic studies, conducted over several years and with appropriate experimental manipulation of variables, of the impact of fire on a single taxon. There are also critical reviews of the fire literature, a discussion of the behaviour of fire in heathland, and several examinations of fire and its effect on insectivorous vertebrates. Both wildfire and prescribed burns are discussed in many of these articles.

Perhaps the most important individual work on the interaction between fire and invertebrates in the RFA area is the unpublished Masters thesis by Dolva (1993). Dolva addressed the effects of litter-reduction burning on the nemobiine gryllid *Nambungia balyarta* in jarrah forest 65 km south of Perth. Dolva concluded that the effect of fire on the cricket's population dynamics could persist for several years after the fire event. This researcher also found that crickets survive best where there is high litter depth, high litter cover and high canopy thickness. Moreover, not only do crickets on unburnt sites have a greater average length, but the genotypes of crickets may also differ between specimens from burnt and unburnt sites.

Laboratory experiments carried out by Dolva revealed that moisture content of its surrounds affects the growth, maturation, potential fecundity and survival of *Nambungia balyarta*. Finally, she warned that removal of the forest litter layer and shrub and herb understorey was not only deleterious to species like *Nambungia balyarta* in terms of loss of shelter and food, but could lead to changes in ecosystem processes. Vulnerable invertebrate species may not be able to adapt to these changes.

The other literature has not approached the issue of fire at the individual invertebrate taxon level. Nonetheless, while most research has been broadly based, individual group studies include those on ants (Boardman 1985; Majer 1977), spiders (Main 1987, 1991; 1995; Strehlow 1993) and termite guilds (Abensperg-Traun and Milewski 1995; Abensperg-Traun, Steven and Atkins 1996). There are several papers on soil invertebrates (McNamara 1955; Springett 1976; 1979). The remainder of the studies provide an overall treatment of invertebrate taxa. In three instances the main focus is on insectivorous mammal and bird species (Hindmarsh and Majer 1977; Sawle 1979; Wooller and Calver 1988), and in one case the emphasis is on the effect of acridids (as plant grazers) on seedling survival after bush-fires (Whelan and Main 1979).

(i) Ants: Of the more specific papers, that by Majer (1977), has already been discussed in part with reference to plantations and mining. The burnt plot sampled by Majer had an ant species richness similar to that of the unburnt and dieback-infected plots. Boardman's (1985) Graduate Diploma project represents an intensive study of the effect of different regimes of controlled burns on ants in jarrah forest. The study was conducted at Dwellingup and Karragullen, and extended over approximately one year. Ants were collected in pitfall traps at monthly intervals. The data showed some variability, but this researcher concluded that all of the plots where prescribed burns had been conducted exhibited differences in abundance, diversity and evenness compared with the control plots. These differences gradually decreased over approximately one year. However, some differences persisted up to three years after the burn. Boardman suggested that evidence supported the fire-relatedness of many of the effects. There was also evidence that an ant community may be less resilient after a hot autumn burn than a cooler autumn burn. Ant evenness was reduced in the hot autumn burn.

(ii) Spiders: In an ecological report on spiders, Main (1987) proposed that relic groups of spiders, and others with poor powers of dispersal, cannot readily re-establish in areas disturbed by felling, burning, clearing and agriculture (even if rehabilitated). Gondwanan relics (e.g. Orsolobidae, Archaeidae, Anapidae and Symphytognathoidea) pre-date fire as

a natural phenomenon, and are therefore endangered by fire as a management tool, unless time of burning coincides with dormancy in the taxa. High risk areas are those with relics of *Nothofagus* dominated associations, such as the tingle forest at Nornalup and Walpole, and south coast heath. A taxonomic study on the occurrence of the mygalomorph genus *Moggridgea* (Main 1991) suggests that the Western Australian species (*M. tingle*) is threatened by controlled burns. Main (1995), in her short study of two mygalomorph spider species in sandplain heath/shrubland, demonstrated that life-history had a influence on a species' persistence after fire: the web-weaver (*Cethegus* sp.) was killed by fire, but recruitment of aerially dispersed spiderlings occurred from surrounding unburnt vegetation, whereas the non-web weaver *Anidiops villosus* had the capacity to survive the fire event, but succumbed in the post-fire period due to inadequate shade and litter, and possibly reduced prey availability and exposure to predators. Juvenile recruitment did not occur in this case.

Strehlow's (1993) BSc (Hons) thesis examined the impact of high intensity autumn fire on ground dwelling spiders in semi-arid shrublands at Durokoppin Reserve, 32 km north of Kellerberrin. In the study area she found 88 species from 31 families. Most of these were vagrant ground-dwelling spiders. A single high intensity control burn was carried out in 1989. Regular sampling of the area by means of pitfall traps, in two control and two impact sites, was conducted over a five-year period. The sampling commenced two years prior to the fire (i.e. in 1987) and concluded two years after the fire (i.e. in 1991). The fire resulted in an immediate reduction in spider abundance and species richness. Recolonization of the burnt area occurred at a rapid rate, with spider populations returning to pre-fire conditions after two years. The arachnid community in 1991 was different in both burn and control sites. Strehlow suggested that climate was an important factor influencing spider communities.

(iii) Termites: Abensperg-Traun and Milewski (1995) surveyed termite species in Fitzgerald River National Park, on the south coast of Western Australia, two years after an intense wildfire burnt large sectors of the Park. They discovered that wood-eating

termites were significantly less abundant and diverse in burnt stands. Abensperg-Traun and Milewski deduced that the insects had perished in the fire rather than succumbing to food limitation. Hard, protective clay mounds and site flexibility of nesting enhanced persistence. Abensperg-Traun, Steven and Atkins (1996) sampled 20 termitaria (mounds), belonging to the harvester termite species *Drepanotermes tamminensis*, in unburnt and adjacent burnt mallee-heath at Yorkrakine, 15 km north of Tammin. They found 40% of termitaria in the burnt area had been abandoned, compared with 10% in the burnt stand. No harvested chaff was found in any of the burnt mounds. Ant invasion was higher in the burnt stand. The researchers suggested that high floristic diversity enhances the resilience of harvester termites to fire. In plant species-poor vegetation zones, the co-occurrence of high-intensity fire with drought or livestock grazing pressures may cause long-term effects on harvester termite populations.

(iv) Soil invertebrates: The effect of fire on soil fauna in jarrah forest is discussed in a early paper, that by McNamara (1955), and by Josephine Springett (Springett 1976, 1979). McNamara sampled the soil invertebrates of a site with well-grown jarrah regrowth (trees 55-65 feet in height), and compared them with those from an annually burnt firebreak. He found a significant difference in gross populations of organisms.

In her earlier paper, Springett (1976) reported on the effects of mild burns in a *Pinus pinaster* plantation and in jarrah and karri forests. The sites chosen were Gnangara (pine), Amphion Jarrah forest (near Dwellingup) and Big Brook Karri forest (near Pemberton). Replicated burnt and unburnt plots were available in the pine plantation but not at the other two sites. In the native forests, sampling methods used were cut soil samples and hand-collection. Collection of fauna from burnt and unburnt sites was conducted on the same day. Microbial action only was investigated in the pine plantation. (Springett's 1971 paper, discussed above, covers the effects of fire on pine plantation microfauna.) Springett concluded that in both native forests, species diversity and density are reduced after burning and do not achieve pre-burning levels within a prescribed burning rotation (five years). She warned that prescribed burning on a five to seven year rotation was likely to permanently simplify the litter fauna and flora.

Springett's later paper (Springett 1979) reports on the sampling of meso- and microarthropod faunas of a jarrah forest 19 km east of Dwellingup, slightly more than a year after the forest was given a hot burn. Springett concluded that burning reduced the numbers of arthropods, the proportion of juveniles, and the proportion of fungus feeders present in the microarthropod population. The decomposer activity of the soil and litter was estimated using buried cotton strips. Springett found the rate of decomposition was also reduced in the burnt jarrah forest.

Four important recent papers by G. R. Friend and his associates give case studies of the effect on invertebrates of fire in heath-shrubland areas (Friend and Williams 1996; Little and Friend 1993), or review research methodology on fire and invertebrates (Friend 1995, 1996).

In a three-year study, Friend and Williams (1996) examined the responses of invertebrates to fire in mallee-heath shrublands in the Stirling Range National Park. Two approaches were adopted by the researchers: (1) A space-for-time approach, whereby a variety of mallee-heath shrublands of different post-fire ages were contemporaneously sampled, and (2) A control burn programme. Two 20 ha areas were burnt in spring and autumn, respectively, and a 400 ha area was burnt in autumn. Samples of invertebrates were obtained over 12-18 months before and after these burns. Members of the Orders Coleoptera, Hymenoptera, Diptera, Hemiptera, Orthoptera and Araneae were targeted, but only Coleoptera were identified beyond ordinal level. The latter were keyed out to 'morpho-species'. At the ordinal level, variation in abundances was attributable more to locality, seasonal and annual effects than to fire. Responses among the beetles, however, reflected changes due to fire as well as those attributable to other causes. Families within which post-fire responses could be detected were Elateridae (2 spp.), Mordellidae (2 spp.), Pselaphidae (1 sp.) aleocharine Staphylinidae (2 spp.) and Tenebrionidae (1 spp.). The authors concluded that identification to 'morpho-species' level is essential to evaluate changes due to fire.

The paper by Little and Friend (1993) records research done at Tutanning Nature Reserve, 150 km southeast of Perth. Sixteen pitfall traps were placed in a grid in each of three study sites with a different fire history. Sampling was done in autumn, winter and spring 1988, and in summer 1989. Invertebrates (excluding Collembola) were identified to Order, with ants (Formicidae), adult Coleoptera and Araneae being sorted to 'morpho-species' level. The trends found in this study suggested that fire does not exert a long-term influence on the structure of the invertebrate communities in the kwongan ecosystems.

The two papers by Friend (1995, 1996) reviewed the research methodology in the literature (24 studies), and evaluated use of various taxonomic groups as post-fire bio-indicators. Both papers were critical of shortcomings in experimental design and length of study, suggested little attention had been paid to the experimental approach, especially in the adoption of adequate pre- and post-fire sampling, and mentioned the considerable variation in the sampling methods used. Friend (1995) suggested that suitable potential indicator groups based on previous studies are Araneae, and probably Lepidoptera, Isopoda, Blattodea and Thysanura. Acarina, Collembola, Isoptera, Coleoptera, Diptera and Hymenoptera (ants) show consistent patterns only in certain habitat types. Certain species or taxa within these groups may be good indicators, though at this stage there is insufficient fire ecology data available.

Friend (1996) listed three important factors which must be addressed by those who use invertebrates in fire ecology studies, namely: (1) invertebrate populations are extremely variable in time and space at a very fine scale. This variability is driven by environmental factors. Furthermore, since invertebrate succession does not necessarily return to the pre-fire state, there is no base-line against which to measure long-term change; (2) Invertebrates appear to be less resilient to fire in more mesic environments than in drier woodland or shrubland ecosystems; and (3) Variability in insect populations is very high. Taxonomic analyses should focus on species or 'morpho-species' if they are to provide information on post-fire effects. Otherwise the high level of variability in insect populations will cloud the analysis. Moreover, more systematic sampling, experimental design and analytical methods need to be adopted. The study should be of a long-term nature, with a pre-fire coverage of the same temporal duration as the post-fire coverage.

Two other studies, among those which have focused generally on invertebrates, compare pre- and post-fire data. These are Abbott (1984) and Majer (1984). Abbott evaluated the recovery of fauna after a controlled moderate intensity summer burn. The site selected was 25 km southeast of Dwellingup in prime jarrah forest. Two adjacent plots, each of 2.4 ha, were chosen. One was exposed to the burn, the other was the forest control. Fifty soil and litter cores were collected randomly from each plot on 18 occasions, commencing from one month prior to the burn and continuing until 2 years and 10 months after the burn. Pitfall traps were installed at the same time the cores were taken, and left for seven days on each occasion. From the soil core data Abbott deduced that nine (stated incorrectly in the paper as "ten") taxa (Megascolecidae, Araneae, Isopoda, Isoptera, Dermaptera, Orthoptera (crickets), Coleoptera larvae (larvae and adults), Diptera (larvae) and Formicidae) showed no significant difference in density from just after the fire (i.e. February 1980) until the end of the sampling period. Diplopoda occurred at significantly greater densities in the burnt plot, and Chilopoda, Thysanura and Blattodea at significantly lower densities in the burnt plot. The total number of taxa and the density and biomass of that part of that invertebrate fauna studied did not differ significantly between plots. Similar numbers of taxa were captured in pitfall traps in both plots. Abbott concluded that all but three taxa (i.e. Chilopoda, Thysanura and Blattodea) had recovered in density within three years of the fire.

Abbott (*op. cit.*) stated that his findings were difficult to reconcile with those of McNamara (1955) and Springett (1976), who claimed frequent fires in the jarrah forest reduced density of the fauna in the soil and litter. Abbott was critical of the earlier studies, citing the inappropriate size of the sampling unit, the number of samples taken, the duration of sampling, the lack of statistical analysis, and the absence of evidence that study areas were adequately matched. However, Abbott's own study lacks adequate pre-fire faunal data. Moreover, the bulk of the fauna examined by Springett belonged to groups disregarded by Abbott in his analysis (especially Collembola and Acarina). Thus these two studies cannot properly be compared. Thirdly, the fact that both control and experimental blocks in Abbott's experiment had been burned every 5-7 years since 1955 suggests that the fauna may already have been substantially altered by fire at the time of Abbott's experiment (see Springett 1971). Finally, there are the complexities inherent in interpreting taxonomically lumped data, which have already been discussed (Friend 1996).

Majer's (1984) study also had a pre-fire sampling period of one month. The study site was situated near Karragullen. Two 20x25 m plots were marked out in the study area. One was the experimental plot, the other the forest control. Both the experimental plot and a surrounding buffer zone were exposed to a control burn of variable intensity. Litter was collected from 10 19x19 cm quadrats in each plot every month, from just prior to the burn to about a year after the burn. From the same points, soil cores were also removed. The soil organisms were collected by means of Berlese funnels, and identified to Order. Majer found that the fire had little or no effect on the soil fauna, whilst those substantially affected were from the litter. Moreover, the soil fauna had recovered more fully 13 months after the fire compared with the litter fauna. This study suffers from the same type of defects as that of Abbott, i.e. both control and experimental plots had a recent fire history, and the taxonomic coverage is only at ordinal level, but these deficiencies are frankly acknowledged and examined in the 'Discussion'.

Another study by Majer (1985) has been discussed in part above in relation to Mining. For his research, Majer also sampled invertebrates in wandoo woodlands with varying fire histories. The study was conducted in the Dryandra State forest. Arthropods were collected in pitfall traps and hand sorted from bags of litter. Identification was at the ordinal level. Majer concluded that the numbers of invertebrates in the litter and on the ground were reduced by fire, and this reduction was probably still apparent four years later.

While most of the studies mentioned above have concentrated on terrestrial, soil and litter fauna, those by Whelan, Langedyk and Pashby (1980) and Whelan and Main (1979) include discussion of other groups. The former paper reports on an invertebrate survey following a wildfire at the University of Western Australia's Marsupial Breeding Station near Jandakot, 25 km south of Perth. The researchers used pitfall trapping, beating of *Banksia* trees and also searched the crowns of *Macrozamia riedlei* (zamia) and *Xanthorrhoea preissii* (grasstree) for arthropods. Trapping and searching were done in burnt and unburnt patches of equivalent area and vegetational composition. More animals were found in burnt than in unburnt *X. preissii* foliage, all animals in *M. riedlei* came from burnt plants, but burnt *Banksia* canopies yielded a much poorer fauna than unburnt canopies. The arthropod fauna was richer in burnt than in unburnt patches. The authors concluded that many normally arboreal arthropods are restricted to moving at ground level after fire. This could help explain the increased activity of ants after fire. The paper by Whelan and Main (1979) is primarily concerned with how insect grazers affect the establishment of plant populations on coastal woodland after fire; consequently the discussion mostly falls outside the scope of this report. However, the researchers noted that grasshoppers were absent from large burnt areas for one to two years after fire.

Main (1981), in a book chapter on fire tolerance of heathland animals, has discussed the response of various invertebrate groups to the actual fire front. Those which are vulnerable because of their habitat or behaviour include gall-formers and stem borers, ants inhabiting litter, termites in dry timber, some trapdoor spiders (*Idiosoma* sp., *Aganippe* sp. and *Idiommata* sp. are mentioned) and pyrgomorphid grasshoppers.

The final three case studies I will mention in this part of the review are those involving insectivorous vertebrates and their prey. Hindmarsh and Majer (1977) investigated the effect of prescribed burning on the availability and abundance of forest invertebrates eaten by the mardo (*Antechinus flavipes*). This mammal mainly feeds from the ground and litter layer. Hindmarsh and Majer selected four similar areas of karri forest in the Pemberton area, which exhibited varying stages of fire succession. They did not find Amphipoda, Isoptera, Heteroptera or dipteran larvae in the most recently burnt areas. On the other hand, Pseudoscorpionida, Chilopoda, Blattodea, Dermaptera, non-formicid Hymenoptera and Coleoptera (Staphylinidae and Curculionidae) increased in abundance shortly after the burn. Other groups were reduced in abundance after the burn, but increased again in the post-fire succession. Since many of the invertebrate groups eaten by the mardo had returned to their former abundance within five years, Hindmarsh and Majer concluded that observed delays in mardo recolonization of burnt areas (up to 20 years) are not due to limited food availability.

Like Hindmarsh and Majer (1977), Sawle (1979) also studied the mardo. She found that this marsupial mostly ate members of the insect Orders Blattodea and Coleoptera (adults and larvae) and also Araneae. Unlike the previous authors, she linked the absence of the mardo from recently burnt areas to the dearth of suitable edible arthropods. The final study, that by Wooller and Calver (1988), examined changes in the number and types of small birds during a three year period after a low intensity fire. These birds were mist-netted in the understorey of dry sclerophyll forest, near Manjimup. The authors found the number of prey taxa fell from 12 to six, the birds ate proportionately more ants and fewer beetles, and that the ants eaten after the fire were smaller than those eaten before it.

Perhaps more than for any other disturbance in the RFA area, finding suitable invertebrate taxa which could be useful as future bio-indicators of fire-related disturbance is a difficult proposition. Not only have earlier studies, in particular, been mainly piece-meal affairs involving brief surveys without further monitoring or experimentation, but the preponderance of research has only dealt with the taxonomy of collected specimens at the higher levels (i.e. Order or family). We also find it hard to escape the conclusion that some of the disagreement expressed between various parties (especially on how the soil fauna responds to fire) may reflect conflicting ideologies. Fire has been used in forestry practice in mesic south-west forests for over 40 years, primarily to reduce the dangerous build-up of flammable litter, and hence damaging wild-fires. Only recently has the conservation aspect, especially in relation to invertebrates, started to take on significance. However this may be, Majer's (1984) finding that litter dwelling organisms may suffer more from fire than those deeper in the soil warrants careful consideration. Fire sensitive taxa mentioned in at least two of the papers discussed under the heading of 'Fire' include several such organisms. Among these are Amphipoda, Blattodea, Isoptera (particularly dry-wood termites), Thysanura and some Araneae (notably Mygalomorphae). Other vulnerable groups include some Coleoptera (particularly at the larval stage) and some Orthoptera (i.e. crickets and pyrgomorphid grasshoppers). To these, perhaps, could be added litter-dwelling Chilopoda and Diplopoda. We suggest that select members of these groups should be the future focus of monitoring for adverse fire-related effects on invertebrate populations.

e. ***Phytophthora*** (Figure 8)

There are surprisingly few papers on the interaction between the dieback fungus *Phytophthora cinnamomi* and forest invertebrates. The two major studies, those of Postle, Majer and Bell (1986) and Newell (1997), present conflicting results. The former study was conducted at Dwellingup. Two plots were selected: a 'dieback control' plot and a 'dieback graveyard'. Soil and litter mesofauna were sampled from 10 randomly selected points in each plot, the litter being removed in sealed plastic bags and soil in soil cores. Invertebrates were extracted by means of Berlese funnels, and sorted to Class or Order. Litter decomposition experiments were also conducted, using leaves placed in

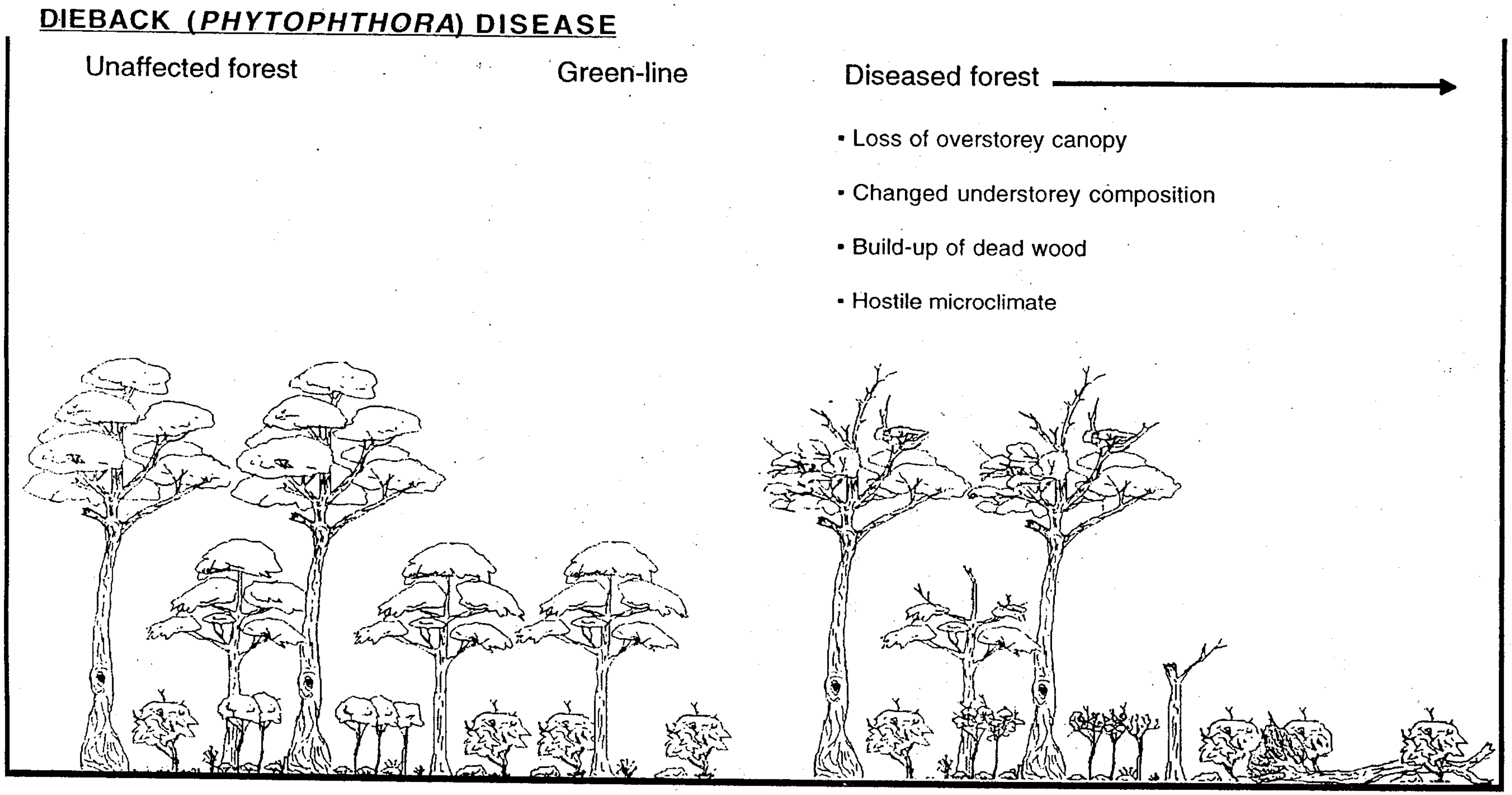


Figure 8. *Phytophthora* attack on jarrah forest vegetation

fine, medium or coarse bags. The researchers found that most taxa were reduced in density or totally absent from the dieback affected plot. The litter decomposition results were equivocal, with most decomposition being observed in the coarse mesh bag. However, this finding could not be positively associated with large soil organisms. The researchers finally concluded that physical factors were responsible for litter decomposing more rapidly in dieback affected plots.

The study conducted by Newell (1997) is included here, although it took place in Victoria. The research was particularly thorough, being part of a broader investigation into the habitat preference of the dasyurid *Antechinus stuartii*, and it is useful to compare the outcome with the West Australian research. Two low open forest sites were selected in the southern part of the Brisbane Ranges National Park, 70 km northwest of Melbourne. The sites were matched as closely as possible for vegetation, aspect and slope. In the first site (termed 'grid A') three groupings of trees were identified (i.e. 'healthy', 'light or recent infection' and 'heavy or long-term infection'), and on the second site ('grid B') there were two uninfected groupings, and one heavily infected grouping. Fifty pitfall traps were placed at regular intervals in each grid, and sampled for one week in each of the four seasons over two years. Samples were grouped into 'macroinvertebrates' or 'microinvertebrates' according to the food preference of *A. stuartii*. Invertebrates were sorted to Order level, except for ants which were identified to 'morpho-species'. Crustacea (Amphipoda, Isopoda and Decapoda), Mollusca (Gastropoda), Annelids (Oligochaeta, Hirudinea) and a number of insect Orders were mostly disregarded in the final analysis because of their low numbers.

The researcher found that significant differences between invertebrate fauna were uncommon and, where identified, the greater abundance figures were on dieback-infected sites. Consistent temporal effects (season and year) were observed in normalised data sets. Abundances, both within individual taxa and pooled counts, were generally weakly associated with ground-level features. Overall, the impact of dieback-infection on vegetative structure and floristics was not reflected in different abundances of terrestrial invertebrates.

The two remaining publications treated here examine dieback-infection as just one among several variables affecting southwest forests. Nichols and Burrows' (1985) study has already been mentioned in relation to mining. The dieback site they examined had a reduced population of predatory invertebrates compared with the healthy forest. Nonetheless, the measured diversity was high, and terrestrial invertebrates were the most severely affected. Nichols and Burrows related this to low leaf litter abundance. Majer (1977) concluded that the dieback-infected forest examined in that study had an ant fauna comprised of an original forest component, and another group of ants commonly associated with disturbed and more open areas. The similarity index for ants collected in the dieback site was midway between that for healthy, unburnt jarrah forest and a burnt, though dieback-free, jarrah forest.

The reasons for apparent discrepancies between data obtained from the Victorian study and the Western Australian studies could be subtle. The Victorian results could even be an artefact resulting from pitfall traps functioning more efficiently in areas where ground vegetation is more open. Differences in such factors as climate, soils, floristics or plant structure may make the two groups of studies difficult to compare. At this stage there is no clear evidence of any taxonomic groups that could be used as bio-indicators of the influence of *Phytophthora cinnamomi* on invertebrate populations.

f. *Agriculture*

Curiously, all the studies on the effects of agriculture on invertebrates in the southwest of Western Australia have concentrated on the central wheatbelt. In fact, all seven of the publications cited here had at least one study site within a small arc commencing from Dowerin in the north and terminating at Kellerberrin in the south! There has been no invertebrate research focusing on agriculture within the jarrah or karri forest areas. Nor have any studies been done on southeastern shrublands or the southwestern or southern coastal belts.

Apart from general coverage of invertebrate groups or subsets thereof, ants (Scougall, Majer and Hobbs 1993), and termites (Abensperg-Traun 1992) have been chosen as target groups. More general studies have examined canopy arthropod faunas (Majer, Recher and Keals 1997), and 'large soil animals' (Abbott, Parker and Sills 1979). Abensperg-Traun, Arnold *et al.* 1996, Abensperg-Traun, Smith *et al.* 1996 and Abensperg-Traun, Smith *et al.* 1997 deal with soil and litter arthropods with some emphasis being placed on ants and, to a lesser degree, termites and several other groupings.

Scougall, Majer and Hobbs (1993) studied the soil, plants and ant fauna of fenced and unfenced remnants of jam wattle (*Acacia acuminata*) and York gum (*Eucalyptus loxophleba*) woodland in the Kellerberrin district. Ants were sampled by pitfall traps only. Trapping was undertaken within a seven-day period in February. Seed removal by ants was also studied over a 24-hour period in March. *Iridomyrmex* spp. were found to be dominant in the non-fenced remnants, whereas generalised myrmicines were more prevalent in fenced remnants. High numbers of native seeds were removed from both fenced and unfenced remnants, with the total amount in the latter being slightly lower and more variable than in the former.

The other specialist paper, that by Abensperg-Traun (1992), examined the effects of grazing on subterranean termites in wandoo woodland 30 km north of Kellerberrin. Sampling was done on five sheep-grazed and five ungrazed plots. Termites were collected from trenches (depth 10 cm, trench volume ≈ 5000 cm^3) and from the inside of partially decayed wood on the soil surface. Sampling was done monthly between August and October. The presence of mounds was also noted. Between 10-15 termite species were collected in grazed plots and 6-14 species in ungrazed plots. Species eating sound wood, wood/debris and grass were equally diverse, and were sampled at similar frequencies in grazed as in ungrazed plots. However, termite mounds were more abundant in grazed plots. Abensperg-Traun concluded that prolonged sheep grazing in wandoo woodland had had neither a detrimental nor a beneficial effect on subterranean termites.

Abbott, Parker and Sills (1979) examined the presence or absence of large soil animals in three sites at York, Corrigin and Kodj-Kodjin (north of Kellerberrin). At each site there was an area of native vegetation separated by a fence and firebreak from farmed or cleared land. An additional site at Kodj-Kodjin had last been cultivated 10 years prior to the study. Sampling was done in winter and summer. On uncleared land, large soil animals (defined as those between 2 and 5 mm; i.e. large enough to create cavities in the soil) were collected from soil cores from under *Eucalyptus* litter. On adjacent cleared land eight soil samples were also taken. Abbott, Parker and Sills found that large soil animals (mainly termites, ants and beetles) were virtually eliminated on regularly cultivated soils. As a result, the cultivated soils, with one exception, were compact and less permeable to water than virgin soil. Large soil animals were almost identical in number in the land cultivated 10 years previously compared with the uncultivated land. In all soils, large soil animals were very scarce in Summer.

Majer, Recher and Keals (1997) studied the effect of habitat fragmentation on canopy arthropods in the Western Australian wheatbelt, and compared their findings with canopy arthropods collected from the Northern Tablelands of New South Wales. In the Western Australian wheatbelt, sampling was performed at three sites: Amery, Namalcatching and Wyalkatchem. At each location, 10 trees were selected for sampling. For standardisation purposes, only wandoo (*Eucalyptus wandoo*) was used. Trees were chosen from four 'growing situations': from the centre of a woodland remnant (remnant centre), from the remnant edge (remnant edge), paddock and corridor. Canopy foliage from trees in the above situations was sampled using a cherry-picker, and sampling was performed during 12 days in September and nine days in November. A total of 60 branchlets were sampled per tree in a 15 minute period. Arthropods were sorted to

Order level, and infrequently collected groups were discarded from the analysis. The authors found that there were significant differences between remnant centres, edges, corridors and paddock trees. Paddocks and edge trees tended to support greater numbers of canopy arthropods than remnant centre trees or those in corridors. Corridor trees supported the fewest arthropods. Arthropod groups where these findings were most noticeable were Homoptera, Araneae and Hymenoptera.

The three papers produced by Abensperg-Traun and his colleagues. (Abensperg-Traun, Arnold *et al.* 1996, Abensperg-Traun, Smith *et al.* 1996 and Abensperg-Traun, Smith *et al.* 1997) report on research done in the same area as Abensperg-Traun (1992). The first study (Abensperg-Traun, Arnold *et al.* 1996) looked at biodiversity in lizards and plants, as well as invertebrates, in gimlet (*Eucalyptus salubris*) woodland and in shrubland. Twenty-seven 0.25 ha sites were established in 18 remnants. Sixteen pitfall traps were installed in each site and trapping was undertaken in woodland in February, April, July and October, and once in shrubland (January). Termites were sampled opportunistically in spring and autumn by the methods described in Abensperg-Traun (1992). Butterflies were sampled only in shrubland, in spring and summer, by means of hand-nets. Arthropods (scorpions, termites, ants, butterflies and adult cockroaches, earwigs, beetles and hemipterans) were identified to 'morpho-species'. The researchers found that in the correlation analyses for woodlands, native plant richness explained 48% of variation of richness in scorpions (with positive correlation), vegetational structure 55% of termites (positive), and litter 59% of beetles (negative). The richness of the shrubland fauna was poorly predicted by the indicator variables (< 25% explained). Various ant functional groups explained 42% each of the richness of scorpions and beetles, and eight beetle species explained 50% of termite richness.

The second study reported on by Abensperg-Traun, Smith *et al.* 1996 was part of the same research programme as that discussed above (i.e. Abensperg-Traun, Arnold *et al.* 1996). However, this study covered the influence of habitat fragmentation and livestock activity on epigaeic arthropod communities in 26 remnants of gimlet woodland. In general, the sampling methods were those described in the other 1996 paper, excepting that butterflies were not collected. Abensperg-Traun, Smith *et al.* found that highly disturbed remnants were associated with; (i) fewer scorpion species, lower termite and ant diversity; (ii) a lower abundance of scorpions, termites and mygalomorph spiders; (iii) more beetle species and higher beetle diversity; and (iv) greater abundance of earwigs and beetles. Cockroach, earwig and ant species richness showed no significant response to disturbance. Species richness of termites, and the abundance of lycosid and idiopid (mygalomorph) spiders, isopods, cockroaches and ants, was highest under moderate disturbance.

The 1997 paper by Abensperg-Traun, Smith *et al.* concludes the reports on the findings of four studies conducted between 1992 and 1997. The faunal aspect of this paper (which also addressed weed invasion and native plant richness and abundance) compared the abundance and species richness of selected soil and litter arthropods in ungrazed and grazed gimlet and wheatbelt wandoo (*Eucalyptus capillosa*) woodland. All disturbed sites for both vegetation types had been grazed intermittently for 30-70 years The sampling methods were the same as those used by Abensperg-Traun 1992, Abensperg-Traun, Arnold *et al.* 1996 and Abensperg-Traun, Smith *et al.* 1996. The researchers found that grazed gimlet woodlands lost a significant number of scorpion and termite species, and these taxa were also significantly reduced in abundance. However, these woodlands supported significantly larger populations of beetles and isopods. Cockroach abundance showed no significant grazing response in either woodland type. Grazed woodlands with the highest levels of weed invasion were associated with the most significant change in arthropod species richness and abundance. Prolonged grazing in the presence of relatively low levels of weed invasion was not associated with significant changes to arthropod species richness and abundance.

The findings discussed above cannot necessarily be extrapolated to the RFA area. However, monitoring of scorpions, termites and mygalomorph spiders in RFA patches close to agricultural land may be fruitful. Additionally, the presence of certain

Iridomyrmex spp. in these areas, especially those species not normally found in pristine forest, and a reduced diversity of generalised myrmicines, may indicate an ant fauna at an early successional stage. This is consistent with a high degree of disturbance. Such a situation could be monitored to see whether there was an eventual return to ant functional group proportions normally associated with the forest.

g. ***Urbanisation*** (Figure 9.)

There is no definitive work on the effects of urbanisation on the invertebrate fauna of the RFA area. What is available are a few papers on ants (Majer and Brown 1986; Burbidge *et al.* 1992), earthworms (Abbott 1982) and termites (Postle and Abbott 1991). There is also a paper and a report analysing results from a survey of Perth remnant bushland reserves (Harvey *et al.* 1997; How *et al.* 1996). All except one of these studies was in the Perth metropolitan area. The only analysis of the effects of clearing for development in a bush setting was conducted at Yanchep National Park (Burbidge *et al.* 1992).

Majer and Brown (1986) carried out a survey of the ant fauna of 33 Perth suburban gardens in 1982. All of the homes involved in the study were within a 20 km radius of the Perth GPO. Ants were sampled by opportunistic hand-collecting during autumn and winter, and by pitfall trapping in December. Ten pitfall traps were inserted in each garden. Forty-seven species from 22 genera were recorded from the gardens. Twenty-four of these (i.e. 51%) were species not recorded in a previous survey of the ant fauna of the Darling Plateau and Swan Coastal Plain. The value of this study is reduced a little by the outdated and occasionally inaccurate taxonomy (e.g. '*Adlerzia froggatti*', a native species, has now been identified as *Tetramorium simillimum*, a cosmopolitan tramp species. This ant was found in 46% of the gardens surveyed.). At least four exotic, tramp species were collected. Large solitary foragers like the bulldog ants (*Myrmecia* spp.) were totally absent. Burbidge *et al.* (1992) has already been discussed under 'Plantations'. The garden sites at Yanchep supported a reasonable range of ant species

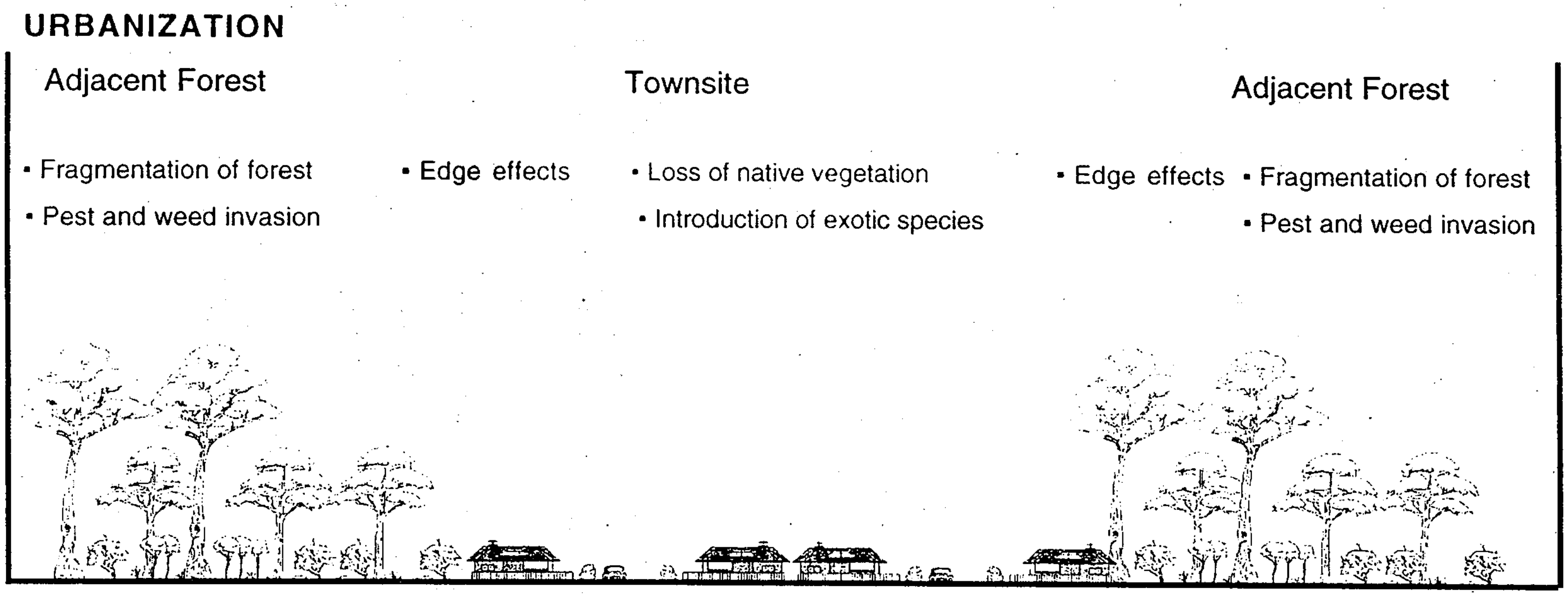


Figure 9. Urbanization

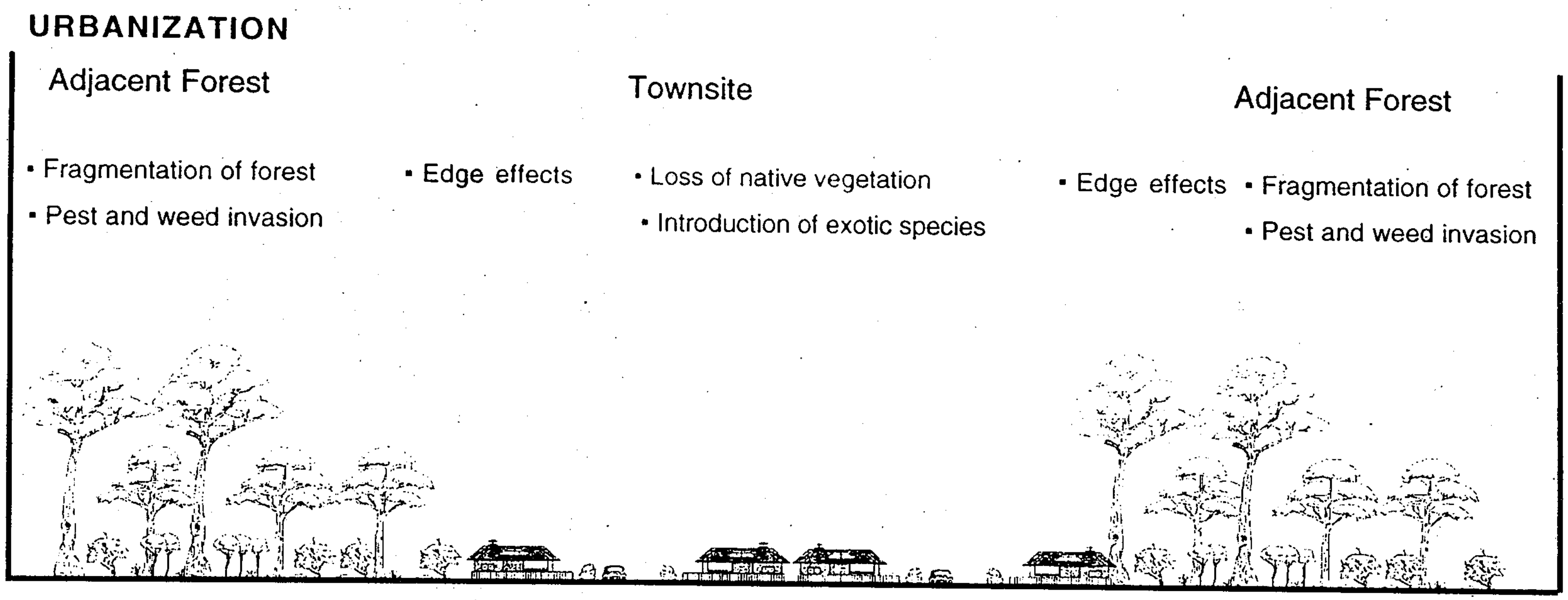


Figure 9. Urbanization

with a high evenness value, i.e. no dominant species were present. Notably, several genera such as *Myrmecia* were present in Yanchep gardens, but absent from those in Perth, suggesting that in Yanchep National Park ants can still recolonise gardens from surrounding bushland.

The paper by Harvey *et al.* (1997) and the report by How *et al.* (1996) report on different stages of the same study programme. The researchers uncovered a rich fauna in Perth remnant bushlands. Arachnida, in particular, collected in this project include several rarities and first records. Harvey *et al.* (*op. cit.*) mentioned that disturbances like fire, weed infestation, grazing and logging have yet to be fully considered in a final analysis of the results of the surveys.

In his natural history survey of the earthworms of the Perth metropolitan area, Abbott (1982) found that all introduced species of earthworms (six out of 13 species) occurred only in sites disturbed by settlement. However, of the seven native species recorded, only two were found in uncleared sites. Abbott surmised that possibly these species also had been introduced from elsewhere. Alternatively, they may be native species that have adapted to land-clearing. In their survey of termites recorded by pest operators in metropolitan Perth, Postle and Abbott (1991) identified four species. Two of these were rare among the material returned by the pest operators, although one (*Nasutitermes exitiosus*) is of economic significance in southern Australia.

The literature is too sparse, and insufficiently experimental in content to enable suitable invertebrate indicators of urban disturbance to be identified at present. Earthworm species may well be better indicators than ants for disturbance surrounding developed areas. However, the taxonomy and biology of native species are poorly known, and these factors, together with the lack of suitably preserved or available type specimens, rule out the use of this group by most researchers.

h. *Recreation*

There is no literature at all on the effects of recreation on invertebrate fauna in the southwest of this state. Undeniably, human activities such as camping, picnicking, trail-bike- and horse-riding, and bushwalking have the capacity to affect invertebrate populations, not only through the trampling of the ground surface and low-lying vegetation, but through the deposition of food-scraps, litter and other forms of waste. Such activities can also introduce exotic organisms, including invertebrate disease pathogens. As an adjunct to the opening up of RFA areas to human recreational activities, there is also pressure on authorities to build support facilities (e.g. toilets, barbecues, stopover huts, safety fences and steps). Tramp ant species, in particular, can be introduced to a pristine area by this means, One of us (BH) has noted how colonies of the Coastal brown ant (*Pheidole megacephala*) have gained access to areas occupied by native ant species in Mt. Coot-tha Forest Park, Brisbane, via the construction of concrete steps for use by walkers in the Park. The lack of research on this area of disturbance should be addressed as a matter of urgency.

i. *Damming*

The only paper on this form of disturbance is that by Woodroff and Majer (1981). These researchers sampled plants ants and other epigaeic invertebrates on the banks of Canning Dam reservoir, which had become exposed as a result of several years of drought. They found that 17 ant species had colonised the reservoir banks within 67 months of exposure. Only three of the ant species collected were not also found in the adjacent forest. Most were generalist feeders that nested in soil or subterranean dead wood. The species composition of ants was very similar to those recolonising bauxite mined areas. Of the other invertebrates, the larger arachnid predators (Scorpionida and Opilionida) were restricted to the upper zones. Furthermore, spiders present in the lower zones were minute. Decomposer invertebrates (e.g. Acarina, some Coleoptera, Collembola and Gryllidae) increased in abundance with increasing time.

j. *Introduced species*

Outside of the domain of economic entomology, ants (notably the argentine ant *Linepithema humile*) are the only introduced invertebrate species in the RFA area covered in the literature. The argentine ant has been implicated in the loss of indigenous ant faunas (Majer 1994). However, our view is that ants are far from being the greatest threat to the forest: phytophagous species that attack the actual trees or understorey and those which transmit pathogens (e.g. some borers and the sirex wood wasp) pose a much greater risk. The argentine ant, and the other major ant pest in this part of Australia, the Coastal brown ant (*Pheidole megacephala*), are primarily inhabitants of towns and disturbed areas close to towns. Judging from the literature, the argentine ant is evidently regarded much more seriously as an invader of outlying disturbed areas. Since the banning of heptachlor in February 1988, this pest has spread rapidly throughout the Perth metropolitan area (WA Agriculture records, BH pers. obs.). Rural shires and towns in or near the RFA area recording *Linepithema humile* include: Albany, Bridgetown-Greenbushes, Bunbury, Busselton, Capel, Chittering, Collie, Cranbrook, Denmark, Donnybrook-Balingup, Harvey, Manjimup, Murray, Narrogin, Plantagenet, Waroona, and West Arthur (Anon. 1988, Figure 2). Nonetheless, in Western Australia this species has not been encountered in undisturbed vegetation (Majer 1994).

WA Agriculture has been monitoring the argentine ant for some years, although this department is no longer responsible for an argentine ant eradication programme. Davis *et al.* (1993) have reported on the influence of this pest on both native ant species and seed removal in the Perth metropolitan and RFA areas. The researchers collected ants by pitfall trapping at three metropolitan lakes, with a total of two infested and two non-infested sites, and at one infested and one non-infested site at Karridale. The sites were subdivided into plots. In the metropolitan localities each site had two plots; 'trees' and 'reeds', respectively. At Karridale sites were located on a farm and in natural bushland. On the farm, each site had two plots, one in a paddock, and the other in an isolated remnant of native vegetation. In the natural bushland, one site (two plots) was in an infested area, and the other (also two plots) was in a non-infested area. The seed removal experiment was conducted at Lake Joondalup, one of the lake sites mentioned above. The two sites at Lake Joondalup were infested and non-infested respectively. To assess seed removal, petri-dish seed depots were installed. These contained seeds of *Acacia saligna* (elaiosome-bearing) and *Eucalyptus gomphocephala* (non-elaiosome-bearing). The seeds were left for 24 hrs. Ants that removed seeds were collected, and the number of seeds removed were recorded.

Davis *et al.* (*op. cit.*) found that there was a clear reduction in ant species where the argentine ant occurred. At the two uninfested, Karridale sites 43 species were captured. This was the same as the number of species collected in the uninfested metropolitan sites. However, in the infested sites the Karridale figures were markedly reduced (to 17 species). Only six species of ants (including *Linepithema humile*) were taken at the infested metropolitan sites. At Herdsman Lake, the argentine ant was the only species collected throughout the year. At other sites at Karridale and in the metropolitan area, there were one to several native species. In winter months at Karridale the researchers found that argentine ants tended to move towards tree lines. In the metropolitan sites single specimens of native ants appeared mainly in July when the activity of the introduced species was reduced. Removal rates of the *Acacia* seedlings were slightly higher in the argentine ant infested sites. (71% compared with 68.5%), but removal of *E. gomphocephala* seeds was much higher in the uninfested sites (52.5% compared with 18%). However, *Acacia* seeds were often left within 5 cm of the seed depots by argentine ants.

Davis *et al.* (*op. cit.*) also warned that opening up areas of native vegetation such as National Parks to public access brought with it a great risk of introducing argentine ants.

Where it is present, the argentine ant is a bio-indicator of disturbance. Particularly, it reveals that the ground has possibly been exposed to increased insolation of the soil, changed moisture regimes or the simplification of the native ant community (Majer 1993).

k. *Pest outbreaks*

Outbreaks of native pests in the RFA area have been covered in a number of articles and technical reports. In the main, these have focused on caterpillar pests; the jarrah leafminer, (Abbott 1992a,b, 1993; Abbott, Van Heurck and Burbidge 1993; Abbott *et al.* 1993; Mazanec 1988, 1989), and gumleaf skeletonizer (Mazanec 1980; Abbott 1992a,b, 1993), but there is also one article on the insect pests of tuart (Fox and Curry 1980).

The jarrah leafminer (*Perthida glyphopa*: Incurvariidae) has a defoliating caterpillar stage. Damage caused by the jarrah leafminer was originally observed on the coast in 1914 (Mazanec 1989), but the first serious outbreak took place east of Manjimup about 1960 (Abbott, Van Heurck and Burbidge 1993). Since that time, incidence of jarrah leafminer damage has gradually spread northwards. By 1989 there was evidence that the pest was successfully completing its life cycle in the Avon Valley and the Perth Coastal Plain (Abbott 1992a).The question of whether outbreaks of the leafminer are facilitated by forestry practice or other human activity is controversial. Mazanec (1988) contended that the pest responds positively to forest thinning. This practice permits crown expansion among the remaining trees, and the rapid growth of saplings from underground lignotubers. The result is a large quantity of foliage, suitable for ovipositing females. In fact, any event in an outbreak area that causes an increase in the quantity of young canopy foliage will attract ovipositing females. Prescribed burning in spring, according to Mazanec (1989), brings about the same effect by releasing soil nutrients locked in the litter and removing undergrowth which competes with jarrah.

Abbott, Van Heurck and Burbidge (1993) have directly repudiated Mazanec's proposed mechanism for jarrah leafminer outbreaks. They did not set up a detailed experimental study, but sought to use geographic replication by surveys, utilising official CALM records. Twenty-five investigations were carried out in order to examine three null hypotheses: (1) prescribed low intensity spring burning does not promote infestation by the leafminer or decrease parasitisation of mines; (2) prescribed moderate intensity burning of jarrah forest in autumn has no impact on the population size of jarrah leafminer; and (3) timber harvesting of jarrah (commercial logging or thinning of jarrah stands) has no impact on population size of jarrah leafminer or on parasitisation of its mines. Abbott, Van Heurck and Burbidge concluded that prescribed low intensity spring fires or timber harvesting did not consistently favour jarrah leafminer or deter the activity of its parasitoid. Moderate intensity fire in autumn reduced the density of jarrah leafminer for 18 months. These researchers proposed that the scarcity of extensive crown scorching after the mid 1950s may have facilitated rapid spread of the pest. This spread was curtailed by the prescribed spring burning program of the then Forests Department.

These conclusions do not sit comfortably with all the data contained in the 'Results': for example, the experimental autumn fire for the Collie region resulted in a significantly increased density of leafminer capsules 18 months after the burn, compared with the stand burned in spring. Moreover, some of the experimental conditions are open to question: the number of leaves tagged per plant (five leaves) in investigations A-K seems rather small. Also, in investigations V to X, the trees at either side of the transects were not matched for burning history (spring versus autumn) <u>or</u> for canopy condition (crown scorch absent in the spring burn versus moderate crown scorch present in the autumn burn). The presence of an extensive clearing (200 m) between the trees for at least three of these investigations also brings in an additional variable, since this would be analogous to a disturbance event affecting the outermost trees on at either side (see Mazanec 1988, above).

As well as the dispute over the aetiology of outbreaks of jarrah leafminer, the question of whether Aboriginal burning of the jarrah forest was a regular event is moot. Abbott, Van Heurck and Burbidge (1993) claimed that areas of jarrah forest heavily used by Aborigines, such as campsites near rivers and swamps, may have been burned annually. Main (1987) cited evidence from an early explorer, Grey, who in 1841 suggested that suggests that fire in the jarrah forest near Harvey was absent. Nearby tableland, however, was fired by the Aborigines. Hallam (1975) understood from early historical records that southern jarrah forests were not habitually burnt by Aborigines.

The pest nearest in importance to the jarrah leafminer is also a lepidopteran, the gumleaf skeletonizer (*Uraba lugens* : Noctuidae). This insect only became a problem in 1983, when outbreaks were recorded in jarrah forest between Nannup and Walpole (Abbott 1992a; Abbott, Van Heurck and Burbidge 1993). However, an earlier outbreak has been recorded between Calingiri and Cowaramup (Abbott 1992b). Like the jarrah leafminer, this pest is a leaf defoliator. Again, as with the former species, the reasons for outbreaks are contentious. Mazanec (1980) supported earlier research that indicated the skeletonizer was favoured by logging and other forestry practices. This was denied by Abbott (1992b), who suggested that rainfall was significant.

The only major pest of karri forest is the bullseye borer (*Tryphocaria acanthocera*: Cerambycidae). Bullseye borer is associated with proximity to mature stands (Abbott 1992b). Other indigenous pests recorded for jarrah include a cup-moth (*Doratifera quadriguttata*), the autumn gum moth (*Mnesampla privata*), five weevils (*Oxyops* spp.), pin-hole borer (*Atractocerus kreuslerae*), the bardi (*Phoracantha semipunctata*), two ambrosia beetles (*Xyleborus saxeni*, *X. banksiae*) and the leafblister sawfly (*Phylacteophaga froggatti*) (Abbott 1992a, 1992b).

Fox and Curry (1980) referred to insect damage to tuart at three localities, extending inland from Burns Beach, 25 km north of Perth to inland of Kwinana (1.8 km to 8.5 km from the ocean). They listed insects known to cause damage to flowers, wood and leaves. The tuart bud weevil (*Haplonyx tibialis*) is the most significant damaging insect on blossom, while serious damage in the wood of healthy trees may result from attack by the tuart borer (*Phoracantha impavida*). The latter can cause ring-barking leading to death of the tree. Other damaging wood-eating insects include the stem girdler (*Cryptophasa unipunctata*), cossid moths (*Culama* sp.) and the pin-hole borer (*Atractocerus kreuslerae*). Leaf damage caused by the leaf blister sawfly (*Phylacteophaga froggatti*) was mentioned by Fox and Curry (*op. cit.*) as being particularly severe at Burns Beach. Other pests include spitfires (*Perga* spp.), leaf beetles (*Paropsis* spp.) and Hemiptera (Eurymelidae, Membracidae, Pentatomidae and Psyllidae). Most insect damage surveyed by Fox and Curry was not ascribed to any particular taxon. These researchers, however, suggested that tuart borer damage in the Perth area may have increased at a result of 'environmental changes'. Fire could be considered the main influence leading to degradation of tuart woodlands as a whole.

Endemic insect pests can be good environmental indicators. Such species as the jarrah leaf miner, the gum-leaf skeletonizer, the bullseye-borer and the tuart wood borer are undoubtedly key invertebrate taxa impinging upon stands of jarrah, karri and tuart, respectively. The main problem here, especially as regards jarrah pests, is diverging views on what constitute optimum conditions for outbreaks of these pests: are forestry practices a major contributor or not? The indicator species are not in question: what is in question are the conditions under which these taxa pose a threat to the ecological health of forest in the RFA area

3. Summary of potential bio-indicator invertebrates based on previous research

The following is a list of potential invertebrate bio-indicators for each of the 11 types of disturbance identified for the RFA area. These are not necessarily the best bio-indicators, but they have been judged by ourselves as the most serviceable based on previously conducted research. Work currently in progress includes research on several other groups (see Appendix 2 for current research which has not yet yielded data).

logging: No suitable bio-indicators yet identified.

plantations: Araneae, Collembola, Formicidae, Isoptera, Oligochaeta (potentially).

mining: Araneae, Collembola, Isoptera (several groups), Oligochaeta (potentially).

fire: Amphipoda, Araneae (Mygalomorphae), Blattodea, Coleoptera (select phytophagous and detritivorous species), Isoptera (dry-wood and harvester species), Orthoptera (especially Gryllidae and Pyrgomorphidae), Thysanura. Possibly also Chilopoda and Diplopoda.

Phytophthora: No suitable bio-indicators yet found.

agriculture: Araneae (Mygalomorphae), Formicidae (notably *Iridomyrmex* spp.), Isoptera, Scorpionida.

urbanisation: Oligochaeta (potentially). Much more work to be done.

recreation: No information.

damming: Arachnida (several groups).

introduced species: Tramp ant species (i.e. the argentine ant *Linepithema humile* and the Coastal brown ant *Pheidole megacephala*).

pest outbreaks: Bullseye borer (*Tryphocaria acanthocera*), gumleaf skeletonizer (*Uraba lugens*), jarrah leafminer (*Perthida glyphopa*), tuart wood borer (*Phoracantha impavida*).

Of the groups examined in the literature thus far, Araneae (particularly Mygalomorphae), Formicidae and Isoptera have the broadest indicator potential. However, their general applicability must be accepted with a degree of caution. In regard to at least some major categories of disturbance, ants, in particular, may respond to changing temporal features rather than to the disturbance itself (e.g. see Newell 1997). Consequently, alterations in community structure are short-term, and unlikely to provide helpful information about any serious long-term effects of the disturbance. Terrestrial molluscs are extremely sensitive to all forms of human disturbance, and with one exception, have not persisted in closely settled areas in Perth. They respond to subtle factors in the soil and vegetation, and often have very localised distributions. Unfortunately, they are not amenable to pitfall trapping and are highly cryptic and difficult to detect. Moreover, they are often only active above ground for a few weeks in a year (S. Slack-Smith, pers. comm.).

4. Effect of disturbance type on terrestrial invertebrates in the RFA area

Not all the disturbance types have a uniform effect on invertebrates. Disturbances can even be subdivided in respect of their short- and long-term effects (e.g. the difference between eucalypt and pine plantations, or fire in fire tolerant or fire susceptible forest types). Table 4 is an attempt to quantify disturbance effects on fauna on the RFA area, based on the research so far completed. The assessments are necessarily subjective and coarse-grained, as many additional factors may ameliorate or exacerbate the impact of different types of disturbance (e.g. seasonality, topography, synergisms or combined disturbance effects, and degree of disturbance).

5. The future search for invertebrate bio-indicators in the RFA area

Thus far, most research on invertebrate bio-indicators in the RFA area and surrounding phytogeographic zones has centred on arthropods, particularly those that are conspicuous, abundant, readily sampled, and which seem to demonstrate a measurable succession of species in the face of alteration to their environment. However, we find it necessary to pause and ask: 'What sort of information is a particular bio-indicator providing for researchers?' This can be varied; the taxon or taxa may be rare and localised so that their presence or absence is a matter for concern, they may have a community structure which alters in a measurable and informative manner given different types or levels of perturbation, or they may constitute a forest pest when conditions are suitable. Two processes are involved here. These are inventorying and monitoring. Kremen *et al.* (1993) define 'inventory' as the documentation of the spatial distribution of biological elements, such as populations, species, guilds, communities and ecosystems. On the other hand, 'monitoring' is involved in assessing changes in ecosystem structure, composition, and function in response to natural factors, human disturbance, or management activities over time.

Table 4. Effects of different types of disturbance on terrestrial invertebrates in the RFA area. * short-term (< 1yr) effects, with few if any permanent changes to invertebrate communities; ** moderate effects, medium (1-5yr) to long-term alteration to community structure, loss of vulnerable taxa (e.g. Gondwanan relic spp.); *** severe long-term (> 5yrs), often permanent effects with loss of biodiversity and community simplification, key taxa (e.g. ants, termites) fixed at early successional stages; **** disturbance most severe, usually permanent and irreversible alteration to fauna, most indigenous species lost, to be replaced by cosmopolitan tramp species, or only a few hardy generalists left; ? effects uncertain, insufficient data.

	Disturbance effect on terrestrial invertebrates				
Disturbance Type	Short-term	Moderate	Severe	Most severe	Data lacking
Logging	*				
Pine plantation				* * * *	
Eucalypt plantation					?
Mining		**			
Fire	*	**			
Phytophthora				*	
Agriculture			***		
Urbanisation				* * * *	
Recreation					?
Damming					?
Introduced species	[result of severe to most severe disturbance]				
Pest outbreaks	*				

Both processes have been implicit in the choice of groups previously used as bio-indictors in research performed in the RFA area, but perhaps the process of monitoring has been the more widely adopted. Inventorying requires a familiarity with taxonomic literature that may not be readily accessible or available for many groups of target taxa. Even sorting to 'morpho-species' level is made more complex if juvenile stages have a different morphology to that of the adult, if sexual dimorphism is pronounced, or if morphological castes are present. Nonetheless, if conservation is to be given a similar standing to commercial forestry practice, the role of inventorying must increase: it is simply not enough to know that insect 'biomass' and 'richness' have not been affected by a particular forestry practice or other anthropogenic disturbance if qualitative evaluation of these parameters is lacking. To give an example, the biomass and richness of ants before and after an area has been urbanised may be comparable; however, the species may be (and usually are) quite different. While this situation involving ants may be discernible to a non-specialist (since many ants are relatively large, fairly well-known taxonomically, and often conspicuous in their habits), drastic changes in, say, a mite community may go unnoticed without a proper process of inventorying.

What we propose for consideration here are general criteria for identifying suitable terrestrial invertebrate bio-indicator groups in the RFA area. This discussion is necessarily preliminary, and will require considerable input from various specialists, as well as groups involved in conservation and applied entomology, before it can develop into an adequate and workable framework suitable for integration into forest management policy. For convenience we separate potential bio-indicators into 'inventory' and 'monitoring' components. 'Inventory' species are those taxa which are of special concern to conservationists, University biological departments and other basic researchers. 'Monitoring' species include taxa whose community changes give information of interest to ecologists, consultants and applied entomologists such as forestry staff and Agriculture WA employees.

a. *Inventory*

Consideration 1. The target taxa should be endemic to the RFA area, though their range may extend beyond it. However, taken as a group, they should be found throughout most of the RFA area, and not be highly localised in a small patch.

Consideration 2. The taxa should be sensitive to disturbance, and respond to it in a measurable way.

Consideration 3. Where a population has been destroyed or depauperised by disturbance, recruitment should be from adjoining areas. Depauperised populations should not easily be able to be replenished by organisms coming from well outside the affected area (apterous species, or those which are clumsy fliers, are particularly suitable in this regard).

Consideration 4. In the consideration of fire, which is perhaps the most important disturbance affecting the whole RFA area, the taxa should be adapted to the original mesic environment which is believed to have existed prior to the formation of the present forest communities. Relic species with known or postulated Gondwanan affinities are specially important. Such organisms often survive in highly specialised and vulnerable situations such as moist gullies, in caves or in damp, cool environments in rocky outcrops.

Consideration 5. The taxonomy of a target group or groups should be adequately known, with 'user-friendly' keys available to enable the non-specialist to identify collections to a suitably low taxonomic level (preferably genus or species-group). Knowledge of the approximate natural range of the group members is also desirable, so that changes in distribution patterns can be discerned. This consideration precludes groups which have not been formally revised, or whose distribution in historic times is unknown or uncertain.

b. ***Monitoring***

Consideration 1. In the case of an invertebrate group or community affected by a perturbation, the group or community should undergo a discernible and measurable successional change as a response to that perturbation. The response should be to the perturbation itself, and not to short-term temporal effects (such as season or year).

Consideration 2. The target organisms should be amenable to collection by means that are simple to use, that collect adequate numbers of specimens and which are affordable.

Consideration 3. The demographics of the taxa involved should be such that under normal conditions their numbers are relatively stable, without violent oscillation.

Consideration 4. Taxa which give shelter to taxonomically related organisms are particularly valuable bio-indicators. Their presence or absence can determine the abundance of the other species. A particularly good example is the mound-building termite *Amitermes obeuntis* Silvestri. The termitaria constructed by *A. obeuntis* are particularly favoured by at least seven other termite species (Bunn 1983).

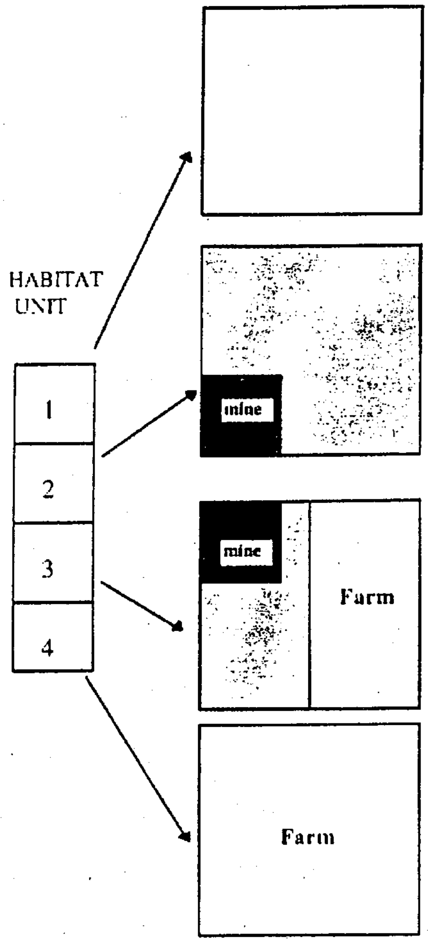
Consideration 5. (NB. Consideration 5 above also applies here).

THE BIODIVERSITY INTEGRITY INDEX

So far we have discussed how individual types of disturbance can affect the invertebrate fauna. It is useful to quantify the extent of the various types of disturbance so that an indication of the magnitude of the impact on animal diversity can be provided. Majer and Beeston (1996) have developed a biodiversity integrity index (BII) which provides a measure of the degree of intactness of the original species richness over a particular land use. It is a product of the species richness of a particular land-use unit and the area which that unit occupies.

Consider a landscape which contains four major habitat units (Figure 10). The units are first standardised to the same area, since the index attributes equal importance to each habitat, whatever the area it occupies. Unit 1 is in a totally pristine state, unit 2 contains a mine which occupies 10% of the area, while unit 3 contains a mine of similar percentage contribution to the area and also farmland covering 45% of the region. The remaining unit has been totally cleared for agricultural purposes. Richness is set at 1 in the pristine state. No weighting is given to habitat units which support exceptionally low or high richness, as equal importance is assigned to the biological diversity of each unit when it is in its pristine state. The richness of a taxonomic group is then assessed within each habitat unit and in each type of disturbance. In the example given, richness drops to a half its original value in the post-mining situation and to a third in the farmland. The BI index in unit 1 is 100 x 1.0, or 100, which is the maximum attainable figure. In unit 2 it is 90 x 1.0 (pristine) plus 10 x 0.5 (mine), which gives a value of 95. In unit 3 the BI index in the disturbed areas is obtained by summing the indices for the pristine area (i.e. 45 x 1.0) and the two types of disturbance (10 x 0.5 (mine) + 45 x 0.33 (farm)), which gives a value of 65, while in unit 4 there are no pristine habitats so the BII is 100 x 0.33, or 33.

The resulting BI indices may be used in a number of ways. First, they can provide numerical data which can contribute towards an audit of the state of biological diversity across the entire landscape. Thus, in the hypothetical region mentioned above, the BI index has dropped from a maximum of 400 down to 293. It is also possible to report on the habitat units where biological diversity is most adversely affected. Thus, the BI index is lowest in unit 4 (33), followed by unit 3 (65). The BI index in unit 2 is only slightly lower (95) than that in the pristine unit 1. An additional use of the procedure is to provide summaries of the impact of different land uses on biological diversity across the entire landscape, or subsets thereof. This information is obtained by subtracting the BII value for a particular land use from that value which would have been obtained had species richness been 1. Thus mining, which occupies 10% of each of two units, has caused a cumulative loss of 10 BII units (10 x 1.0 minus 10 x 0.5 for each unit). Farming, on the other hand, occupies 45 and 100% of two units and has caused a cumulative loss of 30 plus 67, or 97, BI units.



Habitat unit	Proportion of area: Pristine %	Proportion of area: Disturbed %	Proportion of original species in disturbed area	Biodiversity Integrity Index: Pristine	Biodiversity Integrity Index: Disturbed	Biodiversity Integrity Index: Total
1	100	0	n/a	100	0	100
2	90	10	0.5	90	5	95
3	45	(mine) 10 (farm) 45	(mine) 0.5 (farm) 0.33	45 (mine) (farm)	5 15	65
4	0	100	0.33	0	33	33
SUMMARY	235	165	n/a	235	58	293

Figure 10. Diagram of hypothetical region that contains four habitat units which have been subjected to differing amounts of mining and farming, showing how the biodiversity integrity index (BII) is affected by these land uses. The BII value is derived by multiplying the percent area under each land use by the proportion of the original diversity that remains in that land use.

Majer and Beeston (1996) used ants in Western Australia as a demonstration of this index since their distribution is well known and their response to major land uses has been subject to a range of investigations. The vegetation of Western Australia has been mapped by Beard (1990) and, in a course-grain sense, can be reduced to 24 districts or sub districts. The major land uses which have occurred in the State can broadly be described as mining (generally with subsequent restoration), agricultural clearing, rangeland grazing, urbanisation and roadways. The areas of each land use in each phytogeographic district was derived from GIS maps. These were then expressed as percentages of each area and multiplied by the fraction of ant diversity remaining in each land use within that area.

Using this procedure, Majer and Beeston (1996) were able to identify the phytogeographic regions where the BII of ants was most reduced (Figure 11). The resulting rankings closely corresponded with the subjective ordering of areas in the recent Draft State of the Environment (Anon 1997) report. Thus, by considering the ant fauna in this way, an objective means of identifying degraded areas is available which may be used for identifying areas for environmental mitigation. Of interest to this review is the fact that BII in the Warren district (most closely equated to the karri forest) was 80, while in the Dale and Menzies districts (most closely equated to the jarrah forest) it was 66 and 69 respectively. Thus, using ants as the indicator, forest biodiversity in the karri appears to be in a better state than that in the jarrah.

A further spin-off of this procedure is that it provides a statement on the relative impact of different land uses on BII. This may be considered within a particular district or group of districts, or across the whole country or State. The State-wide summary for Western Australia (Figure. 11) indicates that agricultural clearing has had by far the greatest impact on ant BII, closely followed by rangeland grazing. The lower losses of BII as a result of urbanisation, roads and mining puts into context the lower impact of these relatively restricted land uses. The index therefore provides useful contextual information which could be used for prioritising conservation or rehabilitation efforts.

Although the example used ants in Western Australia for deriving the BII, there is no reason why this procedure should not be applied to specific regions of the State, such as the RFA, and to many other plant and animal taxa. The exercise has subsequently been repeated for Western Australia using reptiles (Bracken 1995) and has yielded similar results. However, in order to reliably calculate the fractional changes in richness, it is desirable for the case taxon to be speciose and ubiquitous. It is here that invertebrates are eminently suited to this procedure.

CONCLUSIONS

By now it should be evident that it is desirable to maintain or, in degraded areas encourage the return of, a rich invertebrate fauna containing representatives of the full range of functional groups. But in view of the fact that invertebrate faunas are so diverse, this could be a daunting task. Is it necessary for all of the original invertebrate species to be present or are some of the species redundant in the ecological sense? This is a highly contentious and much-discussed issue (Walker 1992).

Ehrlich and Ehrlich (1981) have likened the situation to rivets on an aeroplane - maybe there is no problem if one or even a few are lost, but there comes a point when so many are missing that the plane crashes. The resilience of an ecosystem, such as an area of rehabilitation or a logged forest, may similarly be affected by an absence of species, but at what point does this reach problematic levels? Walker's (1992) paper on 'Biodiversity and ecological redundancy' was understood by some to suggest that in those functional groups where there was some redundancy, we could afford to lose some of its members. Walker (1995) subsequently clarified this misunderstanding by pointing out that where one of the member species declines or disappears due to species-specific effects, ecological equivalence allows functional compensation by the

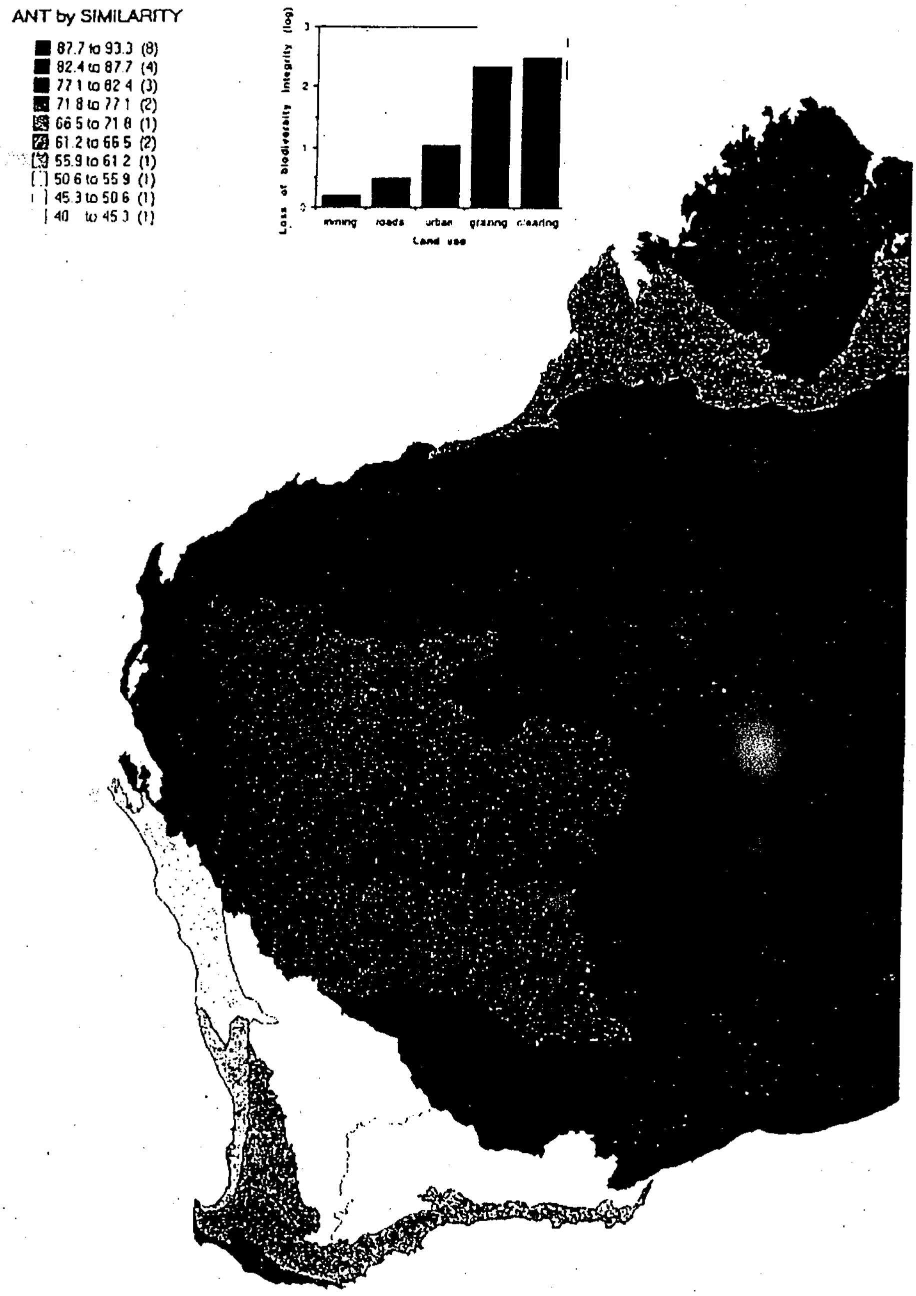


Figure 11. Map of biodiversity integrity (BII) values (based on similarity of ant community in disturbed and undisturbed habitats) in the 24 phytogeographic regions, showing loss of biodiversity integrity(maximum loss is 24 x 100 BI units = 2,400) as a result of five broad land uses in Western Australia (inset). Loss is calculated by subtracting the BII values associated with the five land uses in each region from the values that would have been produced if ant diversity values were still 1.00 (i.e. unaffected) in each of the land uses (data from Majer and Beeston 1996).

other member species that are not so affected. In cases where species in a functional group are virtually equivalent, they probably differ in their environmental adaptations and are thus each able to compensate for species loss under different sets of environmental conditions. We therefore conclude that, wherever possible, land managers should strive to maximise the maintenance or return of the full range of species.

This review of anthropogenic disturbances in forests has indicated the capacity of these operations to alter or eliminate components of the fauna. Some of the practices have greater impacts on the fauna than others, with forest clearing for urban development being at the most adverse end and selective logging being situated more towards the more benign end of the spectrum. However, the extent of the operation must also be taken into account. The total impact on biodiversity is the product of the magnitude of the impact of a particular forestry practice and the area over which it is carried out. Thus a forestry operation which reduces diversity by 10 percent and which is carried out over 90 percent of a forested area can be just as detrimental to the maintenance of biodiversity as a procedure which reduces diversity by 90 percent but which only takes place across 10 percent of the forested area.

We should also mention that this review has only considered the impact of direct exploitation of forests. Pollution, with outcomes such as acid rain, is also seriously affecting the viability of forest animal communities, climate change, which itself partly results from forest exploitation, will increasingly affect the distribution and viability of animal populations world-wide, water damming schemes can inundate entire forests and affect the hydrology of adjacent catchments with flow-on effects to the fauna, mining totally destroys large tracts of forest and associated fauna, which only recovers if adequate rehabilitation is carried out, and of course the effects of firewood cutting, poaching and tourism must also be considered. In addition, there is the multiplier effect of forestry operations on aquatic ecosystems which are affected by erosion and subsequent siltation, a situation which damages river and stream diversity and which creates enormous conflicts between forestry and fishing industries. All of these influences act in tandem with forestry practices to threaten the conservation status of a wide array of animal species.

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Appendix 1 People consulted.

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Ms J. Waldock, West Australian Museum of Natural Science

Dr G. Friend, ex Department of Conservation and Land Management

Appendix 2 Current or planned work which has not yet yielded data

1. **Title**: 'Control of insect pests in young plantations of *Eucalyptus globulus*: Early indicators of pest insect outbreaks and the beneficial impact of spiders and parasitoids.'

Researchers: Abbott, I., Burbidge T., and Wills, A. (Conservation and Land Management [CALM]))

Project: Survey of arthropod faunas on two young plantations of *E. globulus*.

The work was initiated in south-west Western Australian in spring 1993 and continued to late spring 1994. The survey had several goals:

- To document the composition of arthropod fauna on young *E. globulus*, particularly the presence of phytophages of pest potential.
- To test for a relationship between arthropod biomass abundance and tree dimensions (a) as young trees grew during the year, and (b) at a point in time across age classes of trees.
- To identify characteristics in the pest species recovered in the survey, and opportunistically in a wider set of observations, which indicate a potential to affect their distribution and presence within arthropod faunas in *E. globulus* plantations.

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2. **Title**: 'A preliminary study of the soil Acarina (mites) of the soil and litter layers of a *Eucalyptus globulus* plantation in Southwest West. Australia.'

Researcher: H. Adolphson

Degree: MSc.: School of Natural Sciences, Edith Cowan University

This study will compare the mite fauna of a seven-year-old *E. globulus* plantation planted on ex-pastureland, with that of a native jarrah forest selectively clear-felled in 1969 and burned in the spring of 1991. The study is being conducted at Collie.

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3. **Title**: The impact of edges between mature and regrowth karri forest on birds and other invertebrate food resources.

Researcher: P. Atkinson

Degree: PhD: School of Biological and Environmental Sciences, Murdoch University.

Collections of invertebrates have been made in karri and karri/marri forest in Sutton block and Jane block as part of my PhD research into birds and invertebrates and the edges created by logging.

Invertebrates were collected across three height levels using pitfall traps (ground), bush beating (1-2m) and aerial baffle traps (6-10m). Samples were taken across the edges between mature forest and three year old regrowth, mature forest and ten year old regrowth and in mature control forest at least 120m from an edge. Bird census points were positioned at 120m and 60m into regrowth, at the edge, and 60m into mature forest, or at 60m intervals in mature control forest across three replicate edges of each age. There were thus a total of 36 census points, and invertebrates were collected by: nine 5cm diameter pitfall traps, three bush beating samples and two baffle traps each within a 30m diameter area around each census point. Invertebrates were collected in four seasons over one year.

This is work in progress. At this stage, most samples have been sorted to Order. It is intended that all invertebrates collected will be identified to Order and that ants and beetles will be identified to morpho-species. Work on this project is currently on hold, but will recommence in 1998. The data from this project is currently incomplete and not yet available in a useable form. The design of this study ties in with a concurrent study on bird abundance and behaviour in the same areas.

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4. **Title**: The successional response of spider communities to the multiple disturbance of mining and fire in a Western Australian jarrah forest

Researcher: Karl Brennan

Degree: PhD: School of Environmental Biology; Curtin University of Technology

My project will investigate the successional response of spider communities to the multiple disturbances of mining and burning in a Western Australian jarrah forest. This shall be achieved by describing the post-mining succession of spiders recolonising following bauxite removal. Censusing of spiders will be accomplished by sampling a range of sites that represent a chronosequence of increasing ages since mine rehabilitation. the effect of fire on the post-mining spider succession shall be investigated by following the effects of prescribed burns on sites mined five and eight years previously. Censusing of spiders will occur at treatment and control sites both prior to and following the prescribed burns. Sampling of spiders will occur at three-monthly intervals. This method of experimental design will allow any changes in the post-mining spider succession, as a result of fire, to be detected.

My research project shall examine the successional response of spiders to the multiple disturbances of mining and burning in a Western Australian jarrah forest. This shall be achieved by answering the following questions.

(a) What is the successional response of spiders to mining;

(b) How does burning a site mined 5 years previously alter the post-mining spider succession;

(c) How does burning a site mined 8 years previously alter the post-mining spider succession;

(d) How do spider communities at sites mined 5 and 8 years previously differ in their successional response to burning.

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5. **Title**: 'Systematics, biogeography and conservation status of terrestrial isopods (Oniscidea) in south-western Western Australia'.

Researcher: Simon L. Judd.

Degree: PhD: School of Natural Sciences, Edith Cowan University.

The project aims to determine the distribution of terrestrial isopods within the Swan, Jarrah Forest and Warren bioregions of south-western Western Australia. This involves a stratified collecting regime, examination of an extensive collection at the Western Australian Museum, review of the relevant taxonomic and ecological literature and also unpublished literature and private collections should they be available. Assessment of systematics involves reviewing descriptions for species occurring in the area and translating those which are currently not in English. After descriptions and type material are available, collections will be examined in order to review and document the occurrence of existing species and new taxa.

A stratified collecting regime will allow species to be examined in terms of their macro-distribution in relation to rainfall and vegetation type. Detail will also be collected as to microhabitat requirements. This information will be categorised and, when analysed in conjunction with macro-distributional data, should allow for the predictions of selected species' occurrence in particular microhabitats within a region. Armed with such data, replicated field trials will be undertaken to assess the importance of organic matter and the frequency of fire in the maintenance of specific habitat types, and consequently the persistence of some taxa within the landscape. Conclusions will be drawn as to the conservation status of terrestrial isopods and the effect of land management practices upon them.

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6. **Title**: 'Effects of timber harvesting on terrestrial invertebrates inhabiting medium rainfall jarrah forest.'

Researcher: Karin H. Strehlow

Degree: PhD: Murdoch University

The study is being carried out at Kingston, Winnejup and Warrup forest blocks. The study has the following aims:

- To investigate the effects of logging and the associated burning operations on the terrestrial invertebrate fauna.
- To compile information on seasonal abundance, habitat preferences and species composition of the above groups.
- To provide baseline data which will aid in the formulation of general principals for the management of logging in jarrah forest ecosystems.